Reversibility of the effects of acidification and eutrophication of shallow surface waters

Perspectives for restoration

Martijn Bellemakers

Reversibility of the Effects of Acidification and Eutrophication of Shallow Surface Waters

Perspectives for Restoration

Opgedragen aan de nagedachtenis van Johan Bellemakers (1878 - 1960), mijn opa, ontginner en pionier van de Peel, stichter en oprichter van parochie en katholieke openbare school van De Rips en bovenal natuurliefhebber. Uit zijn persoonlijk archief (thans ter bewaring gesteld in het Streekarchief Peelland te Deurne):

Omstreeks september 1934,

"De Rips als natuurschoon monument.

.......Over de paadjes springen en huppelen allerlei kevers en torretjes. Daar ligt een miereleeuw in 'n klein kuiltje in 't zand op zijn prooi te wachten.
Bij hooge uitzondering, ziet men op stille zomeravonden wel 'ns een hert tusschen de hooge boomstammen of 'n schuwe ree. Eenzaam en stil is 't er nu en toch nog vol heerlijke afwisseling, ook nog in deze tijd van 't jaar, nu de bladeren hun herfsttinten beginnen te krijgen. Maar in de lente, als de vogels hun nesten gaan bouwen en hun minnezang gaan zingen, dan is 't vol leven en bonte vroolijkheid.......
En verder, als er twee zandwegen de Peel ingaan, een naar Gemert en een naar Venray, biedt de Rips de natuurliefhebber weer wat anders. Aan de ene kant van de weg weer bosch en aan de andere zijde, 'n onoverzienbare heidevlakte, met een enkel vennetje: 'de Klotterpeel'. Hier heeft de wulp en de kievit zijn broedplaats en in de winter zwerven er wilde eenden en ganzen rond.........
Waar heiblauwtjes in hun kort bestaan van hun leven genieten en vroolijk rondfladderen van struikje tot struikje, gedachtig het woord van de herder: Carpe diem.
Hier en daar kromt 'n pijnboom armzalig omhoog, 'n enkele juniperus wil er nog wel groeien met brem en gagel als buur. De brem o ja, die is er in zijn element, en in de lente staat hij er ook in geel-gouden pracht te bloeien........"

Reversibility of the Effects of Acidification and Eutrophication of Shallow Surface Waters
Perspectives for Restoration

een wetenschappelijke proeve op het gebied
van de Natuurwetenschappen, Wiskunde en Informatica

PROEFSCHRIFT

Ter verkrijging van de graad van doctor aan
de Katholieke Universiteit Nijmegen
volgens besluit van het College van Decanen
In het openbaar te verdedigen
op maandag 9 oktober 2000
des namiddags om 1.30 uur precies

door

MARTINUS JOHANNES SERVATIUS BELLEMAKERS

geboren 16 juni 1960
te Oss

Promotor: Prof. Dr. C. den Hartog

Co-Promotor: Dr. J.G.M. Roelofs

Manuscriptcommissie: Prof. Dr. A.J.M. Smits
Prof. Dr. G. van der Velde (Vrije Universiteit Brussel)
Dr. J.T.A. Verhoeven (Universiteit van Utrecht)

Paranimfen: Germa Verheggen-Kleinheerenbrink
Henk Aarts
Cover: Hanske van Westreenen
Lidwien van der Horst (Grafisch ontwerpster, KUN)
Vormgeving: Janneke Wijgergangs
Tekeningen: Anneke Bellemakers

CIP-DATA KONINKLIJKE BIBLIOTHEEK, DEN HAAG

Bellemakers, Martinus Johannes Servatius

Reversibility of the effects of acidification and eutrophication of shallow
surface waters: Perspectives for restoration / Martinus Johannes Servatius
Bellemakers. -[S.l.: s.n.] (Nijmegen: Mediagroep KUN/UMC).- Ill.
Thesis Nijmegen. - With ref. - With summary in Dutch.
ISBN 90 373 0530 x
Subject headings: acidification / eutrophication / reversibility

Dit proefschrift is gedrukt op nautilus kringlooppapier

CONTENTS

Dankwoord (in Dutch)

Chapter 1:	Introduction	7
Chapter 2:	Effects of liming on water chemistry in shallow acidified pools in the Netherlands: enclosure experiments	19
Chapter 3:	Effects of alkalinity and external sulphate and phosphorus load on water chemistry in enclosures in an eutrophic shallow lake	33
Chapter 4:	Effects of liming on shallow acidified moorland pools: a culture and a seed bank experiment	47
Chapter 5:	Improvement of breeding success of the moor frog (*Rana arvalis*) by liming of acid moorland pools and the consequences of liming for water chemistry and diatoms	63
Chapter 6:	Restoration measures against effects of acidification and eutrophication in shallow surface waters in the Netherlands	77
Chapter 7:	Perspectives for restoration: a synthesis	99
Samenvatting:	Restauratie van verzuurde en geëutrofieerde ondiepe oppervlaktewateren: mogelijkheden tot herstel	119

List of Publications 125

Curriculum Vitae (in Dutch) 128

Dankwoord

Een proefschrift maak je niet alleen. Daarom wil ik iedereen, die op de een of andere manier een bijdrage heeft geleverd aan deze totstandkoming, oprecht bedanken. Een aantal mensen verdienen mijns insziens speciale dankzegging:

- Mario Maessen en Germa Verheggen-Kleinheerenbrink, zij hebben het ruim 6 jaar met me uitgehouden in dat kamertje UL-S35. Herman van Dam voor de fijne en leerzame samenwerking en het goede weer. Volgens het promotiereglement geeft het geen pas om tot promotor en co-promotor bijzondere woorden van dank te richten. Ik wil beiden toch graag bedanken voor de begeleiding van dit onderzoek. Gerard van der Velde voor al het snelle extra correctiewerk. Alle (ex-)collega's van Aquatische Oecologie en alle andere collega's, het is nog maar de vraag of ik ooit nog zulke geanimeerde koffiepauzes met vooral zo vaak taart zal meemaken. Alle studenten die in die tijd hun vakken op "Aquatische" liepen of cursussen/excursies volgden. Zij waren het, die met onbevooroordeelde kijk op zaken mijn gedachten in juiste banen wisten te leiden, door me voortdurend te wijzen op eigenzinnigheden en gebreken.
- Mijn huidige collega's (C&CZ, AEZ en natuurlijk InterNLnet BV), die deze strijd lang aan hebben moeten horen. Vooral mijn collega's van InterNLnet, zij zorgen iedere dag voor gezelligheid, spanning, sensatie en gelukkig nooit voor verveling (zie ook de mailalias fun en http://hetnet.nijmegen.inter.nl.net/personeel/roddelblad).
- Vrienden en bekenden, die uiteindelijk ook niet meer de moed hadden ernaar te vragen. Hele goede vrienden, met wie ik jaren lief en leed deelde en voor wie dit wellicht toch als een verrassing komt.
- Mijn familie (ouders, zussen met aanhang en alle andere familie), peelbewoners van oorsprong. Het viel lang niet altijd mee om uit te leggen wat ik deed en waarom ik dat deed.
- Janneke, Juul en Noortje: ik zal er nooit meer over zeuren...

1

INTRODUCTION

M.J.S. Bellemakers, 2000. Introduction.

Introduction

Shallow surface waters, mainly fed by precipitation, are sensitive to acidification and eutrophication. The major cause for acidification of these waters has been the atmospheric deposition of sulphur oxides (SO_x) and nitrogen oxides (NO_x) (Drabløs and Tollan, 1980; Overrein *et al.*, 1981; Galloway and Dillon, 1983; Schuurkes, 1987). Intensification of agricultural activities in large parts of western Europe has caused an increase of ammonia emissions (Overrein *et al.*, 1981; Roelofs, 1991[a]). Ammonia deposition has led to eutrophication of these waters, in combination with an accelerated acidification owing to nitrification (Galloway and Dillon, 1983; Van Breemen *et al.*, 1984; Schuurkes, 1987).

Weakly buffered, oligotrophic moorland pools

Most of the threatened shallow surface waters of the Netherlands are situated on the pleistocene sandy soils in the southern, eastern, and central parts of the Netherlands and in some coastal dunes. These pleistocene sandy soils are highly weathered and exhibit low acid neutralising capacity (Schuurkes, 1987; Leuven, 1988; Arts, 1990) and thus also the weakly buffered, oligotrophic waters (mainly moorland pools, dune pools and small lakes). Mean values and ranges of water chemical parameters of these surface waters are presented in Table 1. Other physical characteristics for these pools are fluctuating water levels; some pools can even dry up in summer.

Table 1: Mean values and ranges (in brackets) of water chemistry parameters of weakly buffered waters in the Netherlands, classified in three pH-classes (according to Leuven, 1988). Alkalinity is shown in $\mu eq\ l^{-1}$ and the other water chemistry parameters in $\mu mol\ l^{-1}$.

	extremely acid pH < 4.5 n=98		moderately acid 4.5 < pH < 7 n=61		alkaline pH > 7 n=28	
pH	3.9	(3.3-4.5)	5.6	(4.6-6.9)	7.8	(7.0-9.6)
Alkalinity	10	(0-70)	170	(0-720)	790	(120-1900)
Calcium	92	(10-730)	243	(18-1040)	589	(150-1720)
Ammonium	74	(5-305)	73	(4-315)	22	(4-150)
Nitrate	30	(0-775)	39	(2-1450)	161	(1-1659)
Sulphate	257	(0-1280)	345	(50-1180)	463	(67-1580)
Ortho-Phosphate	0.8	(0.1-30.4)	1.8	(0.2-29.8)	1.1	(0.1-4.4)
Chloride	277	(130-740)	475	(160-1794)	889	(205-2500)

The weakly buffered, oligotrophic pools are characterised by communities containing rare and specialised soft water species, such as *Littorella uniflora* (L.) Aschers., *Lobelia dortmanna* L. and *Isoetes echinospora* Durieu (Schaminée *et al.*, 1992). Most species of these characteristic communities are seriously threatened (FLORON Dutch red list species; Weeda *et al.*, 1990) by atmospheric pollution. The soft water macrophytes are mainly water plants with an isoetid growth form (Den Hartog and Van der Velde, 1988) and are adapted to extreme circumstances, due to low levels of carbon dioxide, nitrogen and phosphorus in the water layer (Roelofs *et al.*, 1984). The thick stiff leaves of these macrophytes are compact, and cause a reduction of the surface-volume ratio. Further all isoetids have a well-developed system of internal air lacunae, so that carbon dioxide produced during the respiration can be reused again (Søndergaard, 1979). They are able to take up carbon dioxide with the roots from the sediment pore water, where the carbon dioxide level may be 10-100 times higher than in the overlying water layer (Wium-Andersen, 1971; Søndergaard and Sand-Jensen, 1979; Roelofs, 1983), and they have a high oxygen release by the roots (Sand-Jensen *et al.*, 1982). This can contribute to the oxidation of organic matter and nitrification of ammonium in the sediment, which provides them with carbon dioxide and nitrate.

Furthermore, most of the isoetids apply a special mechanism for photosynthesis, similar to the Crassulacean Acid Metabolism (CAM). This may be interpreted as an adaptation to environments where carbon dioxide availability is precarious (Keeley, 1983). Also, there is a clear correlation between the development of the underground biomass and the nutrient content of the environment; in oligotrophic conditions the root system is well-developed (Sand-Jensen and Søndergaard, 1979). So the isoetids fit functionally very well within the environment where they occur, and this growth form assures the most efficient use of the scarce but essential resources.

Even more, the presence of isoetid species contributes to the maintainance of oligotrophic conditions of soft waters. The extensive root-system aerates the upper sediment layer (Pedersen and Sand-Jensen, 1995). Aeration of the rooting zone stimulates nitrification of ammonia and subsequent denitrification occurs in deeper, anaerobic layers (Risgaard-Petersen & Jensen, 1997). In the aerobic zone, mycorrhizal fungi can grow. Furthermore, the oxygenated upper sediment layer acts as a barrier for phosphate transport from the sediment to the water layer. The oxygen release via the roots is less prominent in other vegetation of rooting soft-water macrophytes (Flessa, 1994).

In environments richer in nutrients the soft-water macrophytes would not stand a chance in the competition with other more demanding species, which usually can grow more efficiently and dominate over the characteristic soft water species (Roelofs *et al.*, 1984; Madsen, 1985; Den Hartog, 1986; Smits *et al.*, 1990).

Acidification of moorland pools

Most of the shallow, oligotrophic, poorly buffered waters have been acidified as a result of atmospheric deposition (Van Dam and Kooyman-van Blokland, 1978; Leuven *et al.*, 1989; Arts, 1990), resulting in a decreased pH of the water layer (Arts, 1990) and raised concentrations of aluminium and sulphate (Schuurkes, 1987; Leuven, 1988). Several decomposition processes become inhibited by low pH, after acidification of the water layer and of the interstitial water of the sediment (Rao and Dutka, 1983). Consequently, a thick, organic sapropelium has developed on the bottom. In many acidified lakes, the transparency increases (Schindler *et al.*, 1985; Yan *et al.*, 1985). As a consequence, the maximum depth of plant growth also increases (Skubinna *et al.*, 1995).
Because of these changes, the characteristic soft water communities have deteriorated (Van Dam *et al.*, 1981; Roelofs, 1983; Leuven, 1988; Arts, 1990). A description of the processes, responsible for the described deterioration is given by Roelofs *et al.* (1984) and Leuven (1988). These studies showed that as a result of acidification, the original vegetation has become overgrown by *Juncus bulbosus* L. and *Sphagnum* spec. (Roelofs, 1983). These submerged species do not have the adaptations of the isoetid macrophytes, but are able to use the temporarily raised carbon dioxide levels of the water layer during acidification more efficiently and can grow faster. *Juncus bulbosus* and *Sphagnum* spec. are more adapted to ammonium uptake than to nitrate uptake, on which many soft water species are dependent (Roelofs *et al.*, 1984). Since the beginning of this century a dramatic decline in the occurrence of the soft water communities has occurred, because they were overgrown by *Juncus bulbosus* and *Sphagnum* spec. (Schoof-Van Pelt, 1973; Roelofs, 1983; Arts, 1990).

Eutrophication of surface waters

Apart from the acidification of moorland pools, many surface waters in the Netherlands have been threatened by eutrophication of the water layer (Van Wirdum, 1979) by precipitation of eutrophicating substances, primarily nitrogen. In these waters the sapropelium layer has increased due to nutrient input. The production of organic matter by algae and submerged macrophytes has exceeded the decomposition. This eutrophication may be caused by an external supply of nutrients (waste water, fertilized (ground)water, atmospheric deposition, bird droppings). The atmospheric ammonia/ ammonium deposition also caused an accelerated acidification of the water layer (Schuurkes, 1987). Eutrophication can also be caused by the release of nutrients by chemical processes in the sediment (internal eutrophication), without external supply of nutrients (Mortimer, 1971; Roelofs, 1991[b]). As a result of eutrophication, the water quality in these surface waters has decreased rapidly (Van Wirdum, 1989; Roelofs, 1991[b]). Eutrophication also threatened the existence of the typical plant communities (Littorelletea; Schaminée *et al.*, 1992). As a consequence of eutrophication, algal blooms can occur and/or the submerged macrophytes become covered by epiphytic algae, reducing the amount of light for the submerged macrophytes (Phillips *et al.*, 1978; Sand-Jensen and Søndergaard, 1979). Pietsch (1994) reports a shift of the species composition within the isoetid community as a consequence of slight eutrophication. With increasing turbidity and epiphytic algal growth, the maximum depth of macrophyte growth decreases (Riis and Sand-Jensen, 1998).

The aim of this thesis

To restore acidified moorland pools and eutrophicated surface waters, the atmospheric deposition of nitrogen and sulphur has to be reduced to values below the critical loads (Bobbink *et al.*, 1992). These critical loads are confirmed by the Dutch government and parliament and laid down by law on resp. 5-10 kg Nitrogen ha^{-1} y^{-1} and 0.3-0.5 kg Sulphur ha^{-1} y^{-1}. This reduction will certainly not be reached before the next 10 years (Cals and Roelofs, 1990; Den Hartog, 1993). Therefore, control measures, which counteract the effects of acidification and eutrophication have been developed, in order to prevent further deterioration of the plant and animal communities of the threatened ecosystems.

The following questions were formulated to obtain a clear idea of the possibilities for restoration of shallow, originally oligotrophic surface waters. These questions formed the base for the research of which the results are presented in this thesis.

- What are the effects of liming the water layer of acidified moorland pools on water chemistry, submerged macrophytes and on the reproduction success of amphibians?
- When is it necessary to remove the organic sapropėlium layer before liming?
- Will inlet of buffered water improve water quality for soft water communities?
- Will a reduction of sulphate and bicarbonate load result in a reduction of the nutrient concentrations of the water layer of shallow eutrophicated surface waters?
- Which control measures give good perspectives for regeneration of the biodiversity of acidified and/or eutrophicated surface waters?

Are restoration measures necessary?

This century, the original natural dynamic processes of most nature reserves on the pleistocene sandy soils in the Netherlands have been diminished dramatically. This can be explained by the small size of these nature reserves (ecological islands), as a result of reclamation of heathlands (marshy lands) from the end of the 19th century until World War II, growing intensification of industrial and agricultural activities (Thissen, 1993) and the fast increase of the population of the Netherlands.

Erosion of inland sandy dune landscapes by wind can create new shallow, weakly buffered, oligotrophic surface waters (*e.g.* moorland pools; Beije *et al.*, 1994). As a result of the diminished dynamics by forestation and agricultural land use and the small size of the Dutch nature reserves this process can no longer take place. Thereby, due to the previously described acidification and eutrophication processes, the existing shallow surface waters have deteriorated (Arts, 1990).
Even when the deposition of acidifying substances should be reduced by 40% between 1986 and 2000, measures against the pollution (atmospheric deposition) are not expected to be effective before the next decade (den Hartog, 1993). After reaching the target loads, many ecosystems will not regenerate spontaneously and need additional measures to restore the original, nowadays threatened or lost biological communities. Therefore, it seems justified to manage these surface waters in a more active way, in order to protect most of the characteristic biological communities of these waters against extinction.

Restoration measures

During this research, the following restoration measures were studied: removal of the organic sapropelium layer and the addition of buffering substances (Figure 1). Regulation of the alkalinity of the water layer is necessary when the natural input of buffering substances has stopped or the input of acidifying pollutants has increased. In acidified ecosystems the alkalinity has decreased, whereas in some eutrophicated waters the alkalinity has exceeded natural levels. The different methods to add buffering substances were liming of the water layer and inlet of buffered surface water.

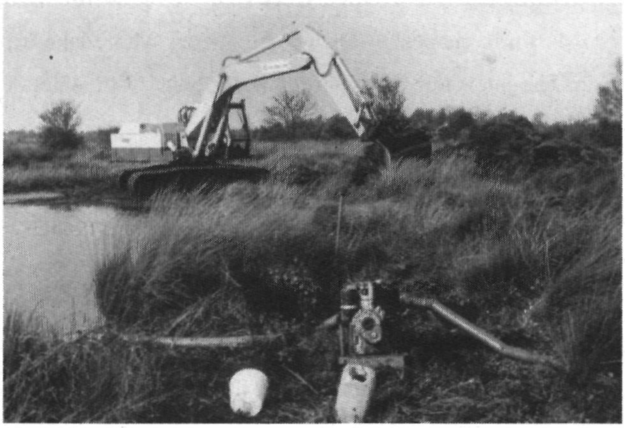


Figure 1: An example of dry removal of the sapropelium layer. In order to keep the pool dry during the work hydraulic pumps are used.

The effects of alkalinity, acidity and external sulphate and phosphorus load on eutrophication processes in the water layer have been studied, in order to gain some insight in the eutrophication processes involved and to develop control measures for the improvement of the water quality.
The final goal of this control management in acidified shallow surface waters was to develop soft waters with relatively low pH-values, low alkalinities and low concentrations of nutrients. After creating these chemical circumstances, it is possible for soft water macrophytes to recover these surface waters (Arts, 1990).

Outline of this thesis

After a short introduction (chapter 1), in the next chapter the effects of liming on water chemistry have been studied. Enclosure experiments have been carried out in two shallow moorland pools before and after removal of the sapropelium layer, in order to understand the effects of liming on water chemistry.
To study the role of sulphate and bicarbonate in the internal eutrophication processes of shallow lakes, enclosure experiments have been conducted in the Naardermeer, a formerly mesotrophic, shallow lake (chapter 3).
In chapter 4 the effects of liming under more experimental circumstances have been studied in an experiment, conducted in the laboratory and in a greenhouse. The effects of the addition of buffering substances on the water chemistry and the effects of liming on the germination and growth of selected submerged macrophytes (*Littorella uniflora* and *Juncus bulbosus*) have been quantified in this study.
In the last 2 chapters of this thesis several whole-lake experiments have been described. The changes in water chemistry, diatom composition and the breeding success of the moor frog (*Rana arvalis* Nilsson) after liming have been shown in chapter 5. The results of four whole-lake experiments have been described in chapter 6. The impact of liming of the water layer or the inlet of buffered water on water chemistry and vegetation development has been evaluated.
The main conclusions of these studies are discussed in chapter 7 and compared with results from the literature.

Acknowledgements

The author is indebted to Drs. M. Maessen, Dr. H. van Dam, Dr. R. Bobbink, Dr. J.G.M. Roelofs and Prof. Dr. C. den Hartog for critically reading and improving the manuscript.

References

Arts, G H P, 1990 Detenoratıon of atlantıc soft-water systems and theır flora, a hıstorıcal account Thesıs Unıversıty of Nıjmegen, 197 pp

Beıje, H M , Van Dam, H and Van der Werf, S , 1994 Heıden, vennen en stuıfzanden In Beıje, H M , Hıgler, L W G , Opdam, P F M , Van Rossum, T A W and Verkaar, H J P A (Eds) Bos- en Natuurbeheer ın Nederland Part 1 Levensgemeenschappen, thırd edıtıon Backhuys Publıshers, Leıden p 217-272 (ın Dutch)

Bobbınk, R , Boxman, D , Fremstad, E , Heıl, G , Houdıjk, A , and Roelofs, J , 1992 Cntıcal loads for nıtrogen eutrophıcatıon of terrestnal and wetland ecosystems based upon changes ın vegetatıon and fauna In Grennfelt, P and Thornelof, E (eds) Cntıcal loads for Nıtrogen p 111-159 Nord (mıljorapport) 1992 41, Nordıc Councıl of Mınısters, Copenhagen

Cals, M J R and Roelofs, J G M , 1990 Prae-advıes effectgenchte maatregelen tegen verzunng en eutrofienng ın oppervlaktewateren Report Department of Aquatıc Ecology and Bıogeology, Unıversıty of Nıjmegen, by order of the Mınıstry of Agnculture, Nature Conservatıon and Fısheneѕ, 96 pp (ın Dutch)

Den Hartog, C , 1986 The effects of acıd and ammonıum deposıtıon on aquatıc vegetatıons ın The Netherlands Proc 1st Internat Symp Watermılfoıl (*Myrıophyllum spıcatum*) and related Haloragaceae specıes, 1985, Vancouver, B C , p 51-58

Den Hartog, C , 1993 Effectgenchte maatregelen tegen verzurıng en eutrofienng ın natuurterreınen In Cals, M J R , De Graaf, M C C and Roelofs, J G M (Eds), Effectgenchte Maatregelen tegen Verzunng en Eutrofienng ın Natuurterreınen, Unıversıty of Nıjmegen, p 1-5 (ın Dutch)

Den Hartog, C and Van der Velde, G , 1988 Structural aspects of aquatıc plant communıtıes In J J Symoens (ed) Vegetatıon of ınland waters Handbook of Vegetatıon Scıence 15 113-153

Drabløs, D and Tollan, A (eds), 1980 Ecologıcal ımpact of acıd precıpıtatıon Proc Int Conf Ecol Impact Acıd Precıp , Norway 1980, SNSF-project, Oslo-Aas 1980, 283 pp

Flessa, H , 1994 Plant-ınduced changes ın redox-potentıal of the rhızospheres of the submerged vascular macrophytes *Myrıophyllum vertıcıllatum* L and *Ranunculus cırcınatus* L Aquat Bot 34 119-129

Galloway, J N and Dıllon, P J , 1983 Effects of acıd deposıtıon the ımportance of nıtrogen In Ecologıcal effects of acıd deposıtıon Natıonal Swedısh Envıronment Protectıon Board, Report PM 1636 146-160

Keeley, J E , 1983 Dark CO_2-fixatıon and dıurnal malıc acıd fluctuatıons ın the submerged aquatıc *Isoetes storkıı* Oecologıa 48 322-333

Leuven, R S E W , 1988 Impact of acıdıficatıon on aquatıc ecosystems ın the Netherlands wıth emphasıs on structural and functıonal changes Thesıs Unıversıty of Nıjmegen, 181 pp

Leuven, R S E W , Van der Velde, G and Kersten, H L M , 1989 Interrelatıons between pH and other physıco-chemıcal factors of Dutch soft waters Arch Hydrobıol 126 27-51

Madsen, T V , 1985 A communıty of submerged aquatıc CAM-plants ın Lake Kalgaard, Denmark Aquat Bot 23 97-108

Mortımer, C H , 1971 Chemıcal exchanges between sedıments and water ın the Great Lakes, speculatıons on probable regulatory mechanısms Lımnol Oceanogr 16 387-404

Overrein, L N , Seip H M and Tollan, A (eds), 1981 Acid precipitation - Effects on forest and fish Final report SNSF-project 1972-1980 Oslo-Aas 175 pp

Pedersen, O and Sand-Jensen, K , 1995 Diel pulses of O_2 and CO_2 in sandy lake sediments inhabited by *Lobelia dortmanna* Ecology 76 1536-1545

Pietsch, W H O , 1994 *Isoetes azorica* Durieu Milde - ein Endemit der Azoren - Vegetations - und Standortsverhältnisse, Gefährdung und Schutzmaßnahmen Phytocoenologia 24 649-665

Phillips, G L , Eminson, D F and Moss, B , 1978 A mechanism to account for macrophyte decline in progressively eutrophicated freshwaters Aquat Bot 4 103-126

Rao, S S and Dutka, B J , 1983 Influence of acid precipitation on bacterial populations in lakes Hydrobiologia 98 153-157

Riis, T and Sand-Jensen, K , 1998 Development of vegetation and environmental conditions in an oligotrophic Danish lake over 40 years Freshwat Biol 40 123- 134

Risgaard-Petersen, N and Jensen, K , 1997 Nitrification and denitrification in the rhizosphere of the aquatic macrophyte *Lobelia dortmanna* L Limnol Oceanogr 42 529- 537

Roelofs, J G M , 1983 Impact of acidification and eutrophication on macrophyte communities in soft waters in the Netherlands I Field observations Aquat Bot 17 139-155

Roelofs, J G M , 1991[a] Vegetation under chemical stress effects of acicification, eutrophication and alkalinisation Thesis University of Nijmegen, 167 pp

Roelofs, J G M , 1991[b] Inlet of alkaline river water into peaty lowlands effects on water quality and *Stratiotes aloides* L stands Aquat Bot 39 267-293

Roelofs, J G M , Schuurkes J A A R and Smits, A J M , 1984 Impact of acidification and eutrophication on macrophyte communities in soft waters II Experimental studies Aquat Bot , 18 389-411

Sand-Jensen, K and Søndergaard, M , 1979 Distribution and quantitative development of aquatic macrophytes in relation to sediment characteristics in oligotrophic Lake Kalgaard, Denmark Freshwat Biol 9 1-11

Sand-Jensen, K , Prahl, C and Stokholm, M , 1982 Oxygen release from roots of submerged aquatic macrophytes Oikos 38 349-359

Schaminée, J H J , Westhoff, V and Arts, G H P , 1992 Die Strandlinggesellschaften (Littorelletea Br -Bl et Tx 43) der Niederlande, in europaischen Rahmen gefasst Phytocoenologia 20 529-558

Schindler, D W , Mills, K H , Malley, D F , Findlay, D L , Shearer, J A , Davies, I J , Turner, M A , Linsey, G A and Cruikshank, D R , 1985 Long-term Ecosystem stress The effects of Years of experimental acidification on a small lake Science 228 1395- 1401

Schoof - Van Pelt, M M , 1973 Littorelletea a study of the vegetation of some amphiphytic communities of Western Europe Thesis University of Nijmegen, 216 pp

Schuurkes, J A A R , 1987 Acidification of surface waters by atmospheric deposition Thesis University of Nijmegen, 160 pp

Skubinna, J P , Coon, T G and Batterson, T R , 1995 Increased abundance and depth of submersed macrophytes in response to decreased turbidity in Saginaw bay, Lake Huron J Great Lakes Res 21 476-488

Smits, A J M , Laan, P, Thier, R H and Van der Velde, G , 1990 Root aerenchyma, oxygen leakage patterns and alcoholic fermentation ability of the roots of some nymphaeid and isoetid macrophytes in relation to the sediment type of their habitat Aquat Bot 38 3-17

Søndergaard, M , 1979 Light and dark respiration and the effect of the lacunal system on refixation of CO_2 in submerged aquatic plants Aquat Bot 6 269-283

Søndergaard, M and Sand-Jensen, K , 1979 Carbon uptake by leaves and roots of *Littorella uniflora* (L) Aschers Aquat Bot 6 1-12

Thissen, P H M , 1993 Heideontginning en modernisering, in het bijzonder in drie Brabantse Peelgemeenten 1850-1940 Thesis University of Nijmegen, Publisher Stichting Matrijs, Utrecht, 331 pp

Van Breemen, N , Driscoll, C T and Mulder, J , 1984 Acidic deposition and internal proton sources in acidification of soils and water Nature 307 599-604

Van Dam, H and Kooyman-van Blokland, H , 1978 Man-made changes in some Dutch moorland pools, as reflected by historical and recent data about diatoms and macrophytes Int Revue ges Hydrobiol 63 587-607

Van Dam, H , Suurmond, E and Ter Braak, C J F, 1981 Impact of acidification on diatoms and chemistry of Dutch moorland pools Hydrobiologia 83 425-459

Van Wirdum, G , 1979 Dynamic aspects of trophic gradients in a mire complex Proc and Inf 25 CHO-TNO, s-Gravenhage p 108-128

Van Wirdum, G , 1989 Ecohydrologische aspecten van waterinlaat in laagvenen in J G M Roelofs (Ed), Aanvoer van Gebiedsvreemd Water Omvang en Effecten op Oecosystemen Laboratory of Aquatic Ecology and Biogeology, Nijmegen, p 52-71 (in Dutch)

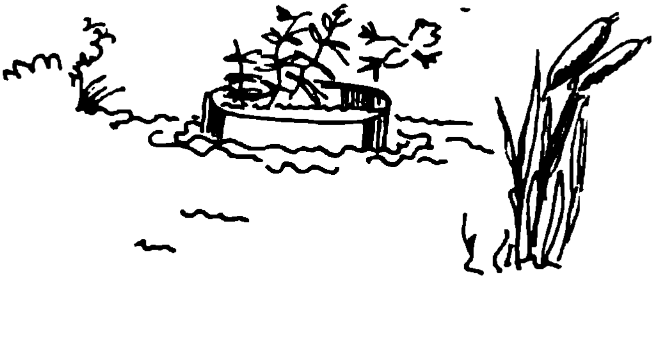
Weeda, E J , Van der Meijden, R and Bakker, P A , 1990 Rode lijst van de in Nederland verdwenen en bedreigde planten (Pteridophyta en Spermatophyta) over de periode 1 1 1980 - 1 1 1990 Gorteria 16 2-26 (in Dutch)

Wium-Andersen, S , 1971 Photosynthetic uptake of free CO_2 by the roots of *Lobelia dortmanna* Physiol Plant 25 245-248

Yan, N D , Miller, G E , Wile, I and Hitchin, G G , 1985 Richness of aquatic macrophyte floras of soft water lakes of differing pH and trace metal content in Ontario, Canada Aquat Bot 23 27-40

2

Effects of Liming on Water Chemistry in Shallow Acidified Pools in the Netherlands
Enclosure Experiments



Abstract

Enclosure experiments have been carried out in two shallow acidified moorland pools in order to study the effects of liming on the water chemistry The addition of buffering substances (sodium bicarbonate and calcium chloride or powdered marlstone) to enclosures in the Ven bij Schaijk, an oligotrophic acidified moorland pool with a mineral sediment, did not demonstrate internal eutrophication After addition of $NaHCO_3$ and $CaCl_2$ the pH and alkalinity increased and all macronutrient concentrations, such as that of phosphate, remained low After treatment with powdered marlstone, there was only a slight increase of pH and alkalinity, due to the slow weathering of marlstone The alkalinity in this moorland pool remained more stable on a mineral sandy substrate than on an organic substrate, probably as a result of a higher acid release from the organic sediment In enclosures in the Padvindersven, an eutrophied, acidified moorland pool with an organic gyttja-type of sediment, internal eutrophication took place after adding buffering substances The phosphate concentration and turbidity of the water increased significantly after treatments with $NaHCO_3$ and $CaCl_2$ as well as with powdered marlstone The acid release was even higher than from the organic sediment from the Ven bij Schaijk It was concluded from these experiments that in case of the Ven bij Schaijk, liming with marlstone would be a sufficient way to restore the original water chemistry In the Padvindersven, however, recovery of the non acidified poorly buffered conditions is only possible by liming in combination with the removal of the organic top layer of the sediment

M J S Bellemakers, M Maessen and J G M Roelofs, 1994 Effects of liming on water chemistry in shallow acidified pools in the Netherlands· enclosure experiments Water, Air and Soil Pollution 73 131-142

Introduction

It has been shown that in the recent past poorly, buffered shallow moorland pools in the Netherlands have been acidified by atmospheric deposition (Schuurkes, 1986; Leuven, 1988). Most of these pools are hydrologically isolated, and fed by precipitation only. Consequently the water has a low alkalinity, is slightly acid and poor in ions. These moorland pools are naturally inhabited by rare plant and animal communities. Therefore, acidification means a serious threat to their existence, and in most of the pools these communities have deteriorated (Van Dam *et al.*, 1981; Roelofs, 1983; Roelofs *et al.*, 1984 and Arts, 1990).

First of all, further acidification has to be prevented by reducing the acidifying substances in atmospheric deposition. Moreover, it is necessary to investigate the possibility for restoration of the acidified and deteriorated ecosystems to the more original, poorly buffered, oligotrophic situation, by active management. The primary production and trophic state within the system can be regulated by manipulating the limiting factors (N, P and C). Since the atmospheric nitrogen deposition is very high in the Netherlands (5.6 kmol N ha^{-1} y^{-1}; Houdijk and Roelofs, 1991), nitrogen limitation is unlikely. Phosphate availability in the water layer depends on pH and alkalinity and thus on inorganic carbon (Carpenter, 1980). Consequently, limitation of inorganic carbon seems to be the key factor that can be manipulated by *e.g.* liming. The natural chemistry of the water of the moorland pools is characterized by a relatively low pH and moderate alkalinity (pH: 5-6; alkalinity: 200-500 $\mu eq\ l^{-1}$; Van Dam and Buskens, 1993).

Most acidified moorland pools have a thick sapropelium layer, due to decrease of microbial activity (Traaen, 1980; McKinley and Vestal, 1982; Raò and Dutka, 1983; Van Dam and Buskens, 1993). After raising the pH and alkalinity (for example by liming) microbial activity can increase (Kok and van de Laar, 1991). In that case internal eutrophication by release of nutrients from the sediment to the water layer, as observed in lakes by Carpenter (1980) can be expected.

Therefore we investigated the effects of liming on water chemistry in very shallow waters, in particular chemical processes, such as internal eutrophication and reacidification.

Study Sites

Field experiments have been conducted in two small, shallow moorland pools in the southern part of the Netherlands: the Ven bij Schaijk and the Padvindersven. Data on water chemistry and on sediment composition are given in Table 1 and 2 respectively.

The Ven bij Schaijk (51° 45 N and 5° 35 E) has a surface area of 0.7 ha and a maximum depth of about 1.5 m. The bottom consists of a mineral sand, is rather flat and hardly grown over; only in the south-eastern corner a sapropelium layer occurs. Prior to the recent acidification this moorland

Table 1: Geometric means and 95% confidence limits of chemical composition of the water at the study sites during the period July 1987-December 1987. n: numbers of samples; Alkalinity and Acidity expressed as $\mu eq\,l^{-1}$; ions expressed as $\mu mol\,l^{-1}$.

	Ven bij Schaijk		Padvindersven	
n	15		12	
pH	4.3	(4.2-4.4)	4.2	(4.1-4.3)
Alkalinity	0	(0-33)	0	(0-0)
Acidity	443	(403-487)	213	(180-253)
Ammonium	17	(10-28)	94	(64-137)
Nitrate	7	(5-10)	6	(4-9)
Phosphate	0.16	(0.09-0.29)	0.15	(0.08-0.29)
Sulphate	502	(460-547)	303	(272-337)
Chloride	601	(566-640)	360	(285-455)
Magnesium	177	(170-184)	63	(58-68)
Calcium	163	(159-167)	107	(100-114)
Aluminum	44	(40-48)	10	(9-12)
Iron	0.7	(0.5-1.0)	0.5	(0.4-0.6)

pool was oligotrophic, poorly buffered with a mineral sediment. Until 1984 specimens of *Luronium natans* (L.) Rafin. (pers. comm. C. den Hartog) were found. In 1987, after acidification, the south-eastern part of the moorland pool was dominated by *Juncus bulbosus* L. and *Sphagnum* spec.
The Padvindersven, part of the nature reserve De Pannenhoef (50° 32 N, 4° 38 E) has a surface area of 2.5 ha and a maximum depth about 1.1 m. The bottom is covered with a sapropelium layer, with a fine structure of sandy granules (gyttja structure). Until 1957 *Littorella uniflora* (L.) Aschers. was found abundantly in this moorland pool; acidification took place in the 1960s and 1970s (Arts, 1990). In 1987 the moorland pool was overgrown with *Drepanocladus fluitans* (Hedw.) Warnst. and *Juncus bulbosus* L.

Table 2: Redox potential, loss on ignition and nutrient composition of the sediments at the beginning of the experiment. Redox potential expressed as mV; ions expressed as $\mu mol\,g^{-1}$ DW; n=1.

	Ven bij Schaijk	Padvindersven
Redox potential	-100	0
Loss on ignition	1%	13%
Total Nitrogen	150	250
Total Phosphorus	0.6	6
Magnesium	8	27
Calcium	6	12
Potassium	25	45

Materials and Methods

Experimental design

To study the effect of additions of buffering substances on water chemistry of waters under field conditions, transparent polycarbonate enclosures (high: 1.50 m; Ø: 1.00 m) were used during the period July 1987 - December 1987. The water inside the enclosures was isolated from the surrounding water by locating the enclosures 15 cm into the sediment. The water in the enclosures was treated as shown in Table 3. In order to enrich the water in the enclosures with calcium bicarbonate, a solution of sodium bicarbonate, and an equivalent amount of a solution of calcium chloride, were regularly added to maintain an alkalinity in the water layer as shown in Table 3. In order to treat enclosures with powdered marlstone, the equivalent of 2 meq l^{-1} (*e.g.* 0.1 g l^{-1}) bicarbonate was added. The chemical composition of the used marlstone is shown in Table 4. At regular intervals (fortnightly) water samples were taken and analysed in the laboratory.

Table 3: Enclosure experiments

Moorland pool	Sediment	Treatment
Ven bij Schaijk	mineral	control
Ven bij Schaijk	mineral	$NaHCO_3$ (500 µeq l^{-1}) and $CaCl_2$
Ven bij Schaijk	mineral	marlstone
Ven bij Schaijk	organic	control
Ven bij Schaijk	organic	$NaHCO_3$ (500 µeq l^{-1}) and $CaCl_2$
Ven bij Schaijk	organic	marlstone
Padvindersven	organic	control
Padvindersven	organic	$NaHCO_3$ (200 µeq l^{-1}) and $CaCl_2$
Padvindersven	organic	$NaHCO_3$ (500 µeq l^{-1}) and $CaCl_2$
Padvindersven	organic	marlstone

Field and laboratory measurements

Measurements of pH, alkalinity, acidity and turbidity were carried out within a few hours after sampling. pH was measured with a GK2501B combined pH electrode, connected to a Radiometer Copenhagen PHM82 pH/mV meter. Alkalinity was determined by titrating 100 ml of water with 0.01 N HCl down to pH 4.2, whereas acidity was determined by titrating 100 ml of water with

Table 4: The chemical composition of marlstone in % of dry weight (B.G.P Paffen, 1990).

CaO	45.30	SiO_2	11.40
Al_2O_3	1.78	Fe_2O_3	0.68
K_2O	0.46	Na_2O	0.13
SO_3	0.53	MgO	0.98
CO_2, H_2O	37.20	Spores	0.54

0.01 N NaOH up to pH 8.2 (Stumm and Morgan, 1981). The turbidity was determined by means of a Toho Dentan model FN5 turbiditimeter. A part of each sample was passed through a Whatman GF/C filter (1.2 µm). These samples were stored in iodated polyethylene bottles and frozen at -28 °C until chemical analysis.

Samples of the top layer of the under water sediments were collected by means of a PVC tube (length: 20 cm) with an inner diameter of 2 cm. Subsamples of the wet sediment were dried at 105 °C during 24 hours. Organic weight was calculated from loss on ignition (4 hrs at 550 °C). In order to establish the chemical composition of the sediment, 50 mg of dry sediment was digested with concentrated sulphuric acid and peroxide as described by Van Dijk and Roelofs (1987). The redox potential of the sediment was measured in the fresh sediment samples after 24 hrs, using a platinum electrode and a Metrohm type 6.0701.100 calomel reference electrode, connected to a Consort D114 digital pH-mV meter.

Sulphate concentrations were determined gravimetrically, according to Technicon Auto-Analyzer Methodology (1981). Colorimetrical measurements were conducted for chloride according to O'Brien (1962), nitrate/nitrite according to Kamphake *et al.* (1967), total ammonia according to Kempers and Zweers (1986) and phosphate according to Henriksen (1965). Calcium, magnesium, aluminum and iron were analysed with the inductive coupled plasma method using an Instrumentation Laboratory Plasma 2000. The colour of the water was determined by measuring the extinction at 450 nm. To calculate the geometric means and the variances of the chemical data of the pools (Table 1), a log-transformation was used, to obtain a normal distribution for all data. Beside that the 95% confidence limits were calculated. Differences in water chemistry of the treatments were tested by using the paired T-Test (Sokal and Rohlf, 1969).

Results

Water flow, wind, dashing of waves and incidence of light can be influenced by enclosures. A comparison between the control enclosures and the open water showed that the effects of the enclosures on water chemistry were negligible in the Ven bij Schaijk, but quite pronounced differences could be observed in the Padvindersven.

The effects of liming on pH and alkalinity are shown in Figure 1. After adding $NaHCO_3$ and $CaCl_2$ both the pH and alkalinity increased significantly ($p < 0.01$) up to 7-8 and 400-550 µeq l^{-1} respectively. When powdered marlstone was used the pH increased also, but stabilised at approximately pH 6. In the water layer of the enclosures above organic substrate the increase of pH was stronger than above mineral sediment. The addition of marlstone powder to enclosures in the Ven bij Schaijk hardly affected the alkalinity (mean alkalinity: 50 and 75 µeq l^{-1} on mineral and organic sediment

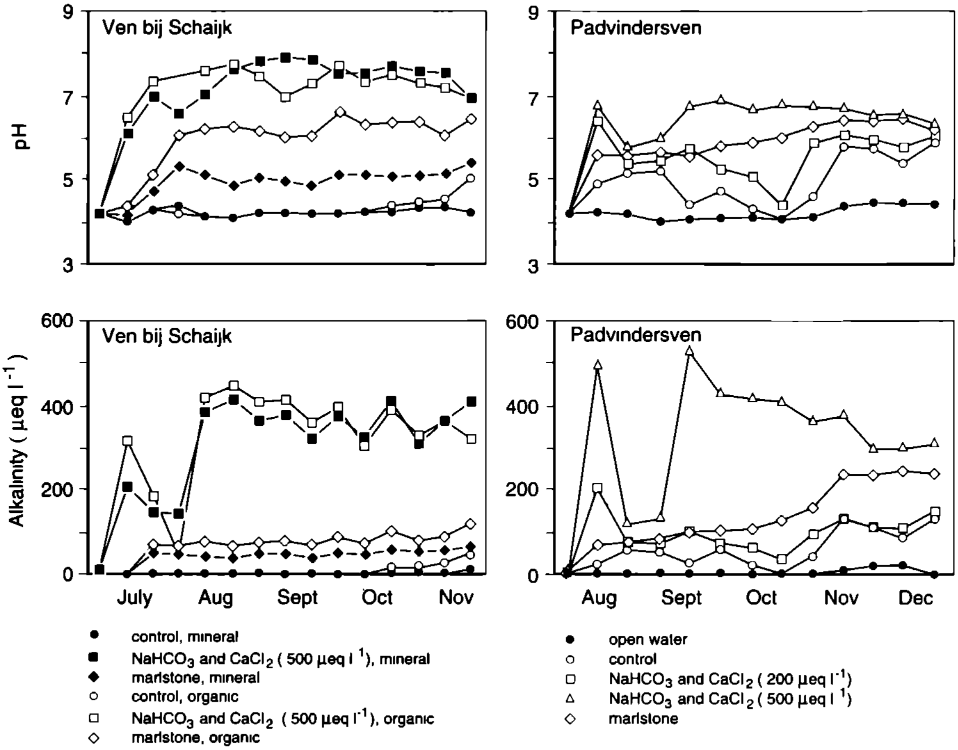


Figure 1: Changes of pH and alkalinity in the enclosures.

respectively). After addition of marlstone powder the rise in alkalinity of the water in enclosures in the Padvindersven was significant compared to the control ($p < 0.01$; mean alkalinity: 140 $\mu eq\ l^{-1}$). After treatment the water layer of the enclosures of the Ven bij Schaijk contained more calcium, but

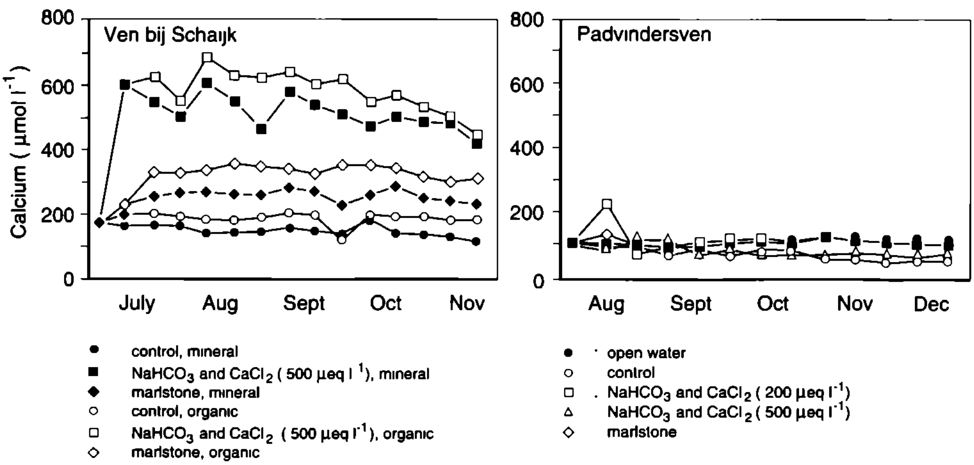


Figure 2: Changes of calcium concentrations in the enclosures.

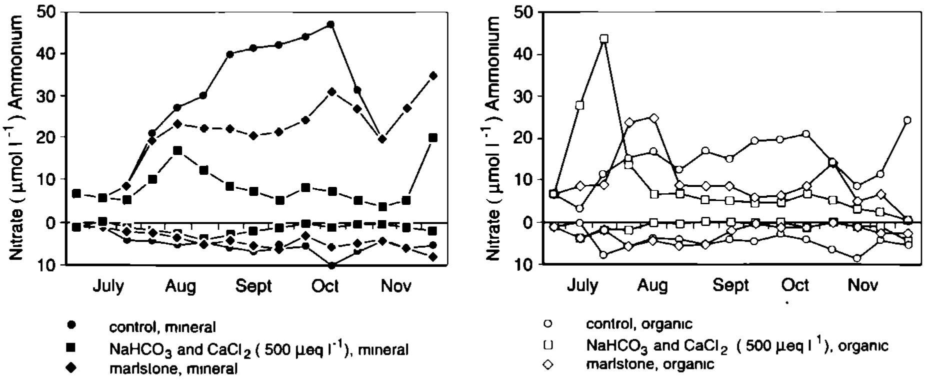


Figure 3: Changes of ammonium and nitrate concentrations in the enclosures of the Ven bij Schaijk.

in the treated enclosures of the Padvindersven the calcium concentrations did not differ significantly from the controls (Figure 2).

The water layer of the Padvindersven contained more ammonium (about 100 to 150 µmol l^{-1}) than that of the Ven bij Schaijk (about 20 µmol l^{-1}; data not shown). In the water layer of the control enclosures on mineral sediment of the Ven bij Schaijk the ammonium concentration in the limed enclosures were lower compared to the control (Figure 3 and 4). However, in the Padvindersven the ammonium concentration was strongly enhanced during the first weeks after treatment with $NaHCO_3$ and $CaCl_2$. At the end of the experiment the ammonium level was about equal compared

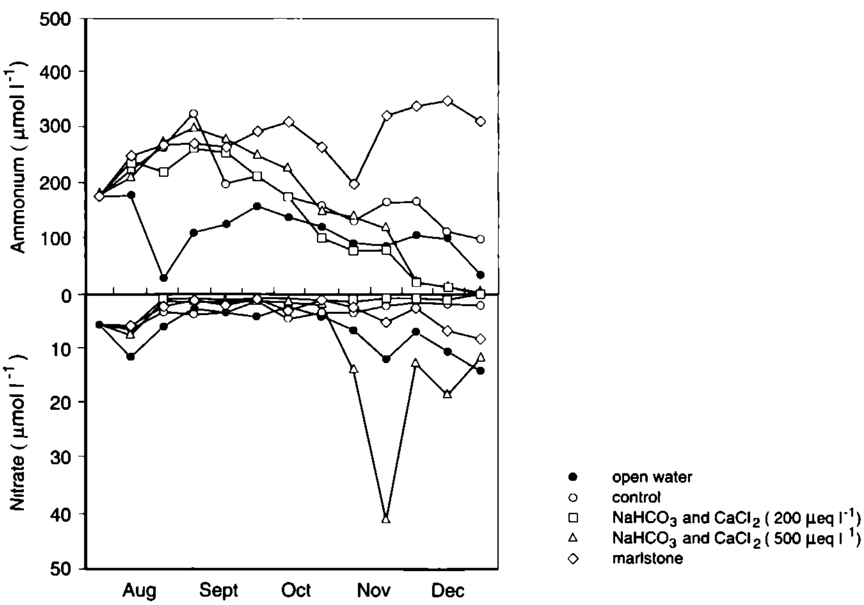


Figure 4: Changes of ammonium and nitrate concentrations in the enclosures of the Padvindersven.

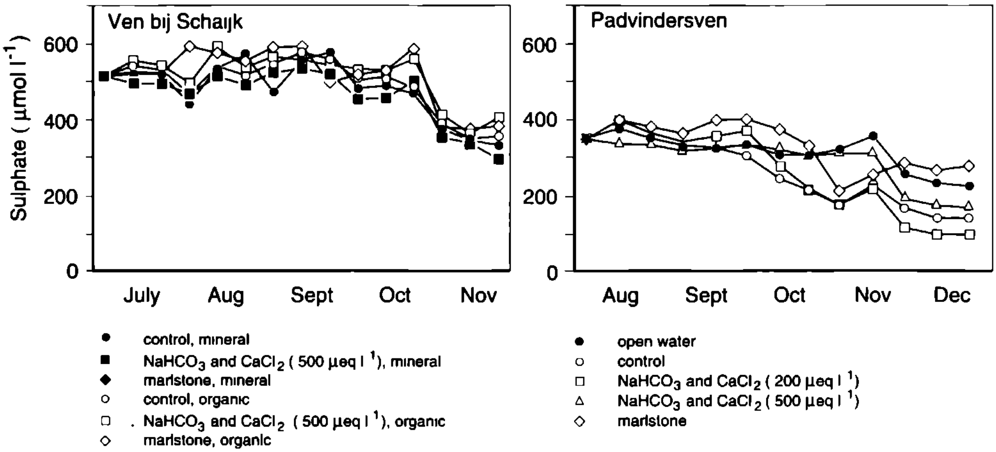


Figure 5: Changes of sulphate concentrations in the enclosures.

to the control. The addition of marlstone powder caused a similar and significant effect, however with a delay of a few weeks ($p < 0.05$).

The nitrate levels seemed to be hardly affected by the various treatments as all nitrate concentrations remained low (Figures 3 and 4).

The time trends of sulphate and chloride are shown in Figure 5 and 6, respectively. During the experiments the sulphate concentrations decreased in the open water and the enclosures of both pools with approximately 200 $\mu mol\ l^{-1}$. As the chloride concentrations did not change in the moorland pools and enclosures, this decrease is not due to dilution by atmospheric precipitation.

The results of the phosphate analysis and turbidity measurements are shown in Figure 7. In the Ven bij Schaijk liming did not significantly affect the phosphate concentration and turbidity. In the Padvindersven the rise of pH and alkalinity resulted in a significant increase ($p < 0.01$) of the phosphate concentrations and turbidity of the water layer.

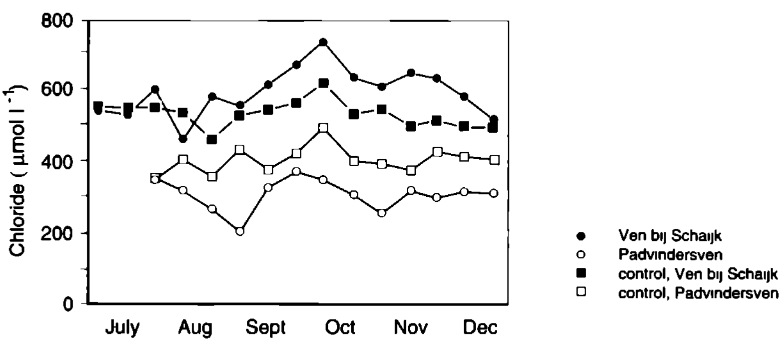


Figure 6: Changes of chloride in the open water and enclosures.

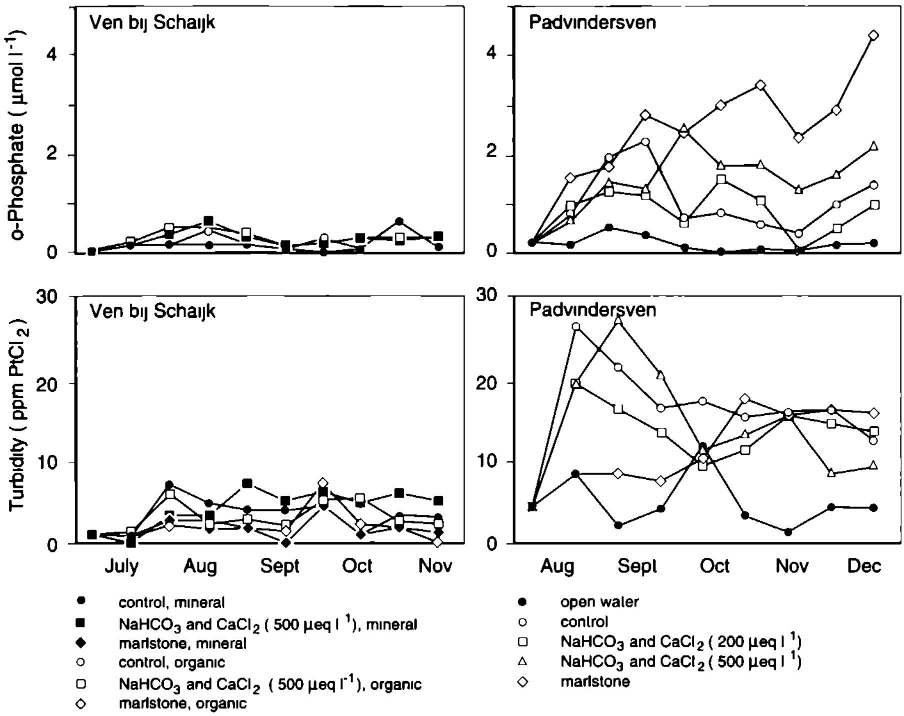


Figure 7: Changes of ortho-phosphate concentrations and turbidity in the enclosures.

Discussion

Enclosure experiments

Enclosure (limnocorral) experiments on the effects of liming of acidified bodies of water have been carried out in the U.S.A. by Bloesch *et al.* (1988). In most of these experiments only the water layer was studied, without the sediment. In the shallow waters of this study the contact between sediment and water layer is much more important, because the sediment can act either as a sink or source of nutrients (Istvánovics *et al.*, 1986), and, therefore, both sediment and water have been considered. In our experiments, we have raised pH and alkalinity and studied the effects on water chemistry. To maintain the alkalinity in the treated enclosures, $NaHCO_3$ and $CaCl_2$ had to be added several times. The decrease of pH was due to acidification by atmospheric deposition (Van Dam *et al.*, 1981; Schuurkes, 1986; Leuven, 1988) and the release of protons by the soil absorption complex as a consequence of the addition of $NaHCO_3$ and $CaCl_2$ (Lindmark, 1982). During the last decades protons have accumulated at the soil absorption complex, as a result of acidification. The release of protons also caused a decrease of alkalinity (i.e. bicarbonate concentration) according to the following reaction: $\mathbf{HCO_3^- + H^+ \longrightarrow H_2CO_3 \longrightarrow H_2O + CO_2}$

In enclosures of the Padvindersven more $NaHCO_3$ and $CaCl_2$ was needed compared to the enclosures of the Ven bij Schaijk, as the release of protons from the sediment to the waterlayer of the Padvindersven was larger than in the Ven bij Schaijk (Bellemakers *et al.*, 1990). An indication for the release of protons is that the calcium concentrations did not increase in the water layer of the enclosures in the Padvindersven after liming with $NaHCO_3$ and $CaCl_2$ or powdered marlstone. Although, there was a slight increase of pH, because the calcium could also be exchanged against other positive ions, such as ammonium and sodium. However, most of the protons were exchanged by the added calcium, causing reacidification of the water layer. Since these processes hardly took place in the Ven bij Schaijk, the calcium concentrations increased after adding $NaHCO_3$ and $CaCl_2$ or powdered marlstone. After the addition of marlstone powder there was only a slight increase of the pH, due to the slow weathering of marlstone.
Several biochemical transformations influence the water chemistry parameters, such as pH, alkalinity, ammonium-, nitrate- and sulphate concentrations. These processes are summarised in Table 5. The accumulation of ammonium in the water layer of the enclosures on mineral sediment (Ven bij Schaijk) was probably caused by atmospheric deposition (Schuurkes *et al.*, 1988[a]). During these experiments the average concentration of ammonium in an enclosure can rise approximately 100 $\mu mol\ l^{-1}$, assuming that there is a deposition of about 1 kmol ammonium ha^{-1} $year^{-1}$ (Houdijk and Roelofs, 1991). In enclosures on organic sediment there is no accumulation of ammonium, probably as a result of plant uptake (*Juncus bulbosus* L.) and/or binding at the soil adsorption complex (Alexander, 1977). Adding powdered marlstone causes the same effect, but with some delay. After liming, the ammonium concentrations decrease strongly, likely due to increased nitrification. The nitrification rate is high under circumneutral low alkaline conditions, while nitrifying bacteria are less active in acidified bodies of water (Rao and Dutka, 1983; Schuurkes *et al.*, 1988[a]). Figure 3 and 4 show that hardly a decrease of ammonium occurs after adding powdered marlstone. The pH in the enclosures remains relatively low (5-6) and the nitrification process appears to be incomplete. Since the nitrate concentrations after treatment hardly increase, it is likely that the nitrate, formed by nitrification, disappeared due to denitrification and/or uptake by plants. Nitrate in the water layer of the enclosures on organic soils disappeared totally by denitrification. This phenomenon was also observed by Knowles (1982) and Schuurkes *et al.* (1988[a]).
The patterns of the sulphate concentrations are clearly related to the presence of organic matter in the soil. This can explain why the sulphate concentrations in the water layer of the Ven bij Schaijk were higher than those in the Padvindersven. During the experiments the sulphate concentrations decreased in both pools. Van Dam (1988) and Schuurkes *et al.* (1988[b]) also reported such a decrease of sulphate above organic sediments. It is likely that this decrease in sulphate concentrations is due

to sulphate reduction (Table 5; Equation no. 5) in the organic sediment, as appears from the measured redox potentials, which are within the range of sulphate reduction (Boström *et al.*, 1982). Sulphate reduction is involved in alkalinization processes which may reduce the acidifying impact of sulphuric acid loads (Table 5; Equation 5ᵃ and 5ᵇ).

Table 5: Biochemical transformations of sulphur and nitrogen

1) Nitrification	: $NH_4^+ + 2\ O_2 \longrightarrow NO_3\ + 2\ H^+ + H_2O$
2) Ammonium uptake	: $NH_4^+ + ROH \longrightarrow RNH_2 + H^+ + H_2O$
3) Denitrification	: $4\ NO_3\ + 5\ CH_2O + 4\ H^+ \longrightarrow 2\ N_2 + 7\ H_2O + 5\ CO_2$
4) Nitrate uptake	: $NO_3^- + ROH + H^+ \longrightarrow RNH_2 + 2\ O_2$
5) Sulphate reduction	: $SO_4^2\ + 2\ CH_2O \longrightarrow S^2\ + 2\ H_2O + 2\ CO_2$
a) At low pH	: $S^2\ + H^+ \longrightarrow HS$
b) At high pH	: $CO_2 + H_2O + S^{2-} \longrightarrow HCO_3\ + HS$
6) Sulphide oxidation	: $H_2S + 2\ O_2 \longrightarrow SO_4^2\ + 2\ H^+$

High ammonium levels in interstitial water are often characteristic for reductive and microbiologically active organic substrate with a high ammonification and low nitrification (Roelofs, 1991). The high ammonium levels in the water layer of the Padvindersven could be explained by the high atmospheric deposition and the reduced nitrification rate only. Also the release of ammonium from the reductive organic soil plays an important role. In general the highest ammonium levels were found in the treated enclosures, probably as a result of an increased cation exchange at the soil adsorption complex caused by increased calcium and sodium levels.

The results of differences in water chemistry in enclosures of the Ven bij Schaijk showed that different treatments did not lead to internal eutrophication of the water layer, independent of the type substrate. The phosphate concentrations remained extremely low and the water very clear. However, the experiments in the Padvindersven showed that the various treatments led to internal eutrophication. The phosphate levels in the water layer increased significantly and also the turbidity of the water was enhanced as a result of algal bloom.

General implications

There are several differences in geomorphology between the two studied moorland pools. Originally the Ven bij Schaijk was a poorly buffered, oligotrophic pool, which has acidified during the last decades. The sediment is still mineral and has a coarse structure. The experiment has shown that the previously described water chemistry characteristics return in enclosures by liming with $NaHCO_3$ and $CaCl_2$ or powdered marlstone, without internal eutrophication. Marlstone seems to be the most suitable as a liming substance. It prevents a pH shock, due to the slow weathering of marlstone.

The Padvindersven is a more eutrophic acidified moorland pool, probably due to a relatively eutrophic situation before acidification The sapropelium layer accumulated far more protons and nutrients than the sediment of the Ven bij Schaijk The sediment also has a more organic and fine structure (gyttja-like) For this reason treatment with lime will facilitate re-acidification and leads to internal eutrophication

Therefore, liming of pools with an organic sapropelium layer will be unsatisfactory if one wants to restore them within a few decades to their original condition Expensive measures, such as removal of the organic sapropelium layer is necessary, to avoid internal eutrophication

Acknowledgements

This research was funded by the Ministry of Housing, Physical Planning and Environment, Directorate Water (Project 011661) We are greatly indebted to Prof Dr C den Hartog, Prof Dr G van der Velde, Dr H van Dam, Dr A W Boxman, and Drs M J R Cals for critically reading the manuscript and to Mrs G M Verheggen and Mr E A J M Gloudemans for technical assistance during chemical analysis Our thanks are extended to the Landscape Foundation of the Province of Noord-Brabant for permission to do research in their properties

References

Alexander, M , 1977 Introduction to soil microbiology Wiley, New York, 467 pp

Arts, G H P, 1990 Deterioration of atlantic soft-water systems and their flora, a historical account Thesis University of Nijmegen, 197 pp

Bellemakers, M J S , Maessen, M and Verheggen, G M , 1990 Restauratie van verzuurde en geeutrofieerde zwak gebufferde ondiepe oppervlaktewateren mogelijkheden tot herstel Report Department of Aquatic Ecology and Biogeology, in order by the Ministry of Housing, Physical Planning and Environment, 98 pp (in Dutch)

Bloesch, J , Bossard, P, Buhrer, H , Burgi, H R and Uehlinger, U , 1988 Can results from limnocorral experiments be transferred to in situ conditions? Hydrobiologia 159 297-308

Bostrom, B , Jansson M and Forsberg, C , 1982 Phosphorus release from lake sediments Arch Hydrobiol Beih Ergebn Limnol 18 5-59

Carpenter, S R , 1980 Enrichment of lake Wingra, Winsconsin, by submerged macrophyte decay Ecology 61 1145-1155

Henriksen, A , 1965 An automated method for determining low-level concentrations of phosphate in fresh and saline waters Analyst (London) 90 29-34

Houdijk, A L F M and Roelofs, J G M , 1991 Deposition of acidifying and eutrophicating substances in Dutch forests Acta Bot Neerl 40 245-255

Istvánovics, V , Voros, L , Herodek, S , Tóth L G , and Tátrai, I , 1986 Changes of phosphorus and nitrogen concentration and of phytoplankton in enriched lake enclosures Limnol Oceanogr 31 798-811

Kamphake, L J , Hannah S A and Cohen, J M , 1967 Automated analysis for nitrate by hydrazine reduction Water Res 1 205-206

Kempers, A J and Zweers, A , 1986 Ammonium determination in soil extracts by the salicylate method Comm Soil Sci Plant Anal 17 715-723

Knowles, R , 1982 Denitrification Microbiol Rev 46 43-70

Kok, C J and Van de Laar, B J , 1991 Influence of pH and buffering capacity on the decomposition of *Nymphaea alba* L detritus in laboratory experiments a possible explanation for the inhibition of decomposition at low alkalinity Verh Internat Verein Limnol 24 2689-2692

Leuven, R S E W , 1988 Impact of acidification on aquatic ecosystems in The Netherlands with emphasis on structural and functional changes Thesis University of Nijmegen, 181 pp

Lindmark, G , 1982 Acidified lakes sediment treatment with sodium carbonate - a remedy? Hydrobiologia 92 537-547

McKinley, V L and Vestal, J R , 1982 Effects of acid on plant litter decomposition in an arctic lake Appl Environ Microbiol 43 1188-1195

O'Brien, J , 1962 Automatic analysis of chlorides in sewage wastes Engineering 33 670-672

Paffen, B G P , 1990 "Onderzoek naar de mogelijkheden van hoogveenregeneratie in de Groote Peel" Report Department of Aquatic Ecology and Biogeology, University of Nijmegen, in order of the Ministry of Agriculture, Nature Conservation and Fisheries 117 pp (in Dutch)

Rao, S S and Dutka, B J , 1983 Influence of acid precipitation on bacterial populations in lakes Hydrobiologia 98 153-157

Roelofs, J G M , 1983 Impact of acidification and eutrophication on macrophyte communities in soft waters in the Netherlands I Field observations Aquat Bot 17 139-155

Roelofs, J G M , Schuurkes J A A R and Smits, A J M , 1984 Impact of acidification and eutrophication on macrophyte communities in soft waters II Experimental studies Aquat Bot 18 389-411

Roelofs, J G M , 1991 Inlet of alkaline river water into peaty lowlands effects on water quality and *Stratiotes aloides* L stands Aquat Bot 39 267-293

Schuurkes, J A A R , 1986 Atmospheric ammonium sulphate deposition and its role in the acidification and nitrogen enrichment of poorly buffered aquatic systems Experientia 42 351-357

Schuurkes, J A A R , Jansen J and Maessen, M , 1988[a] Water acidification by addition of ammonium sulphate in sediment water columns and in natural waters Arch Hydrobiol 112 495-516

Schuurkes, J A A R , Kempers A J and Kok, C J , 1988[b] Aspects of biochemical sulphur conversions in sediments of a shallow soft water lake J Freshw Ecol 4 369-381

Sokal, R R and Rohlf, F J , 1981 Biometry, second edition Freeman, San Francisco, 859 pp

Stumm, W and Morgan, J J , 1981 Aquatic chemistry Wiley and Sons, 780 pp

Technicon Auto-Analyzer Methodology, 1981 Industrial Method 635-81W, New York

Traaen, T S , 1980 Effects of acidity on decomposition of organic matter in aquatic environments In D Drabløs and A Tollan (Eds) Ecological Impact of Acid Precipitation Proc Int Symp , March 1980, Sandefjord, Norway 340-341 New York, U S A , 780 pp

Van Dam, H , Suurmond G and ter Braak, C J F , 1981 Impact of acid precipitation on diatoms and chemistry of Dutch moorland pools Hydrobiologia 83 425-459

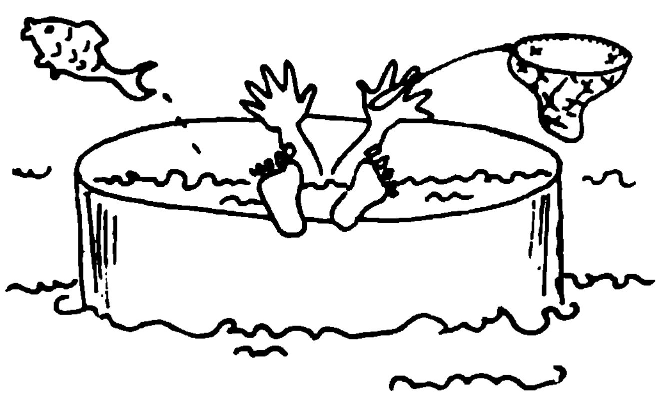
Van Dam, H , 1988 Acidification of three moorland pools in the Netherlands by acid precipitation and extreme drought periods Freshwat Biol 20 157-176

Van Dam, H and Buskens, R F M , 1993 Ecology and management of moorland pools balancing acidification and eutrophication Hydrobiologia 265 225-263

Van Dijk, H F G and Roelofs, J G M , 1987 Effects of excessive ammonium deposition on the nutritional status and condition of pine needles Physiol Plant 73 494-501

3

Effects of Alkalinity and External Sulphate and Phosphorus Load on Water Chemistry in Enclosures in an Eutrophic Shallow Lake



Abstract

As a result of changes in hydrology, the former mesotrophic, shallow lake Naardermeer, has been eutrophicated during the last decades. To compensate for shortage of water, eutrophicated water with different chemical characteristics has been supplied. In order to determine the effects of alkalinity, acidity and external sulphate and phosphorus load on eutrophication processes, developments in water chemistry have been studied in enclosures. A decrease of the phosphorus load of the lake did not improve water quality on the short term. This observation was also confirmed by this enclosure experiment. Reduction of alkalinity did improve water quality, particularly by reductions of turbidity and chlorophyll-a content. The significant increase of the sulphate load in the water layer led to deterioration of the water quality and an increased biomass of sessile algae by internal eutrophication, probably as a result of increased sulphate reduction in the sediment. Therefore, it seems that lowering of sulphate concentrations in the inlet water after the removal of phosphorus is necessary to improve the water chemistry. The best option, however, is to restore the former hydrological conditions after which the system will be fed only by seepage of ground water poor in nutrients and sulphate.

M.J.S. Bellemakers and M. Maessen, 1998. Effects of alkalinity and external sulphate and phosphorus load on water chemistry in enclosures in an eutrophic shallow lake. Water, Air and Soil Polution 101: 3-13.

Introduction

As a result of human activities, such as changed land use and water extraction, most surface waters in the western part of the Netherlands have been eutrophicated during the last decades (Roelofs, 1991). This eutrophication can be caused directly by eutrophic inlet water (rich in nitrogen and phosphorus), that is used to compensate for water losses (external eutrophication; Koerselman and Verhoeven, 1989). However, if the inlet water is poor in nutrients, the macro-ion composition may cause an increased mobilisation of nutrients from the sediment as a result of chemical reactions (internal eutrophication; Mortimer, 1971; Forsberg, 1989; Roelofs, 1991).

Eutrophication leads to an increase of epiphyte growth on submerged macrophytes (Sand-Jensen and Søndergaard, 1981). As a result of shading by epiphytes, the vitality of the macrophytes decreases and more nutrients become available for pelagic algae. This process gradually results in a dominance of phytoplankton, increased turbidity of the water layer and the inhibition of germination and maintenance of submerged macrophytes (Phillips *et al.*, 1978).

An important cause for internal eutrophication is a reduction of seepage, due to increased water extraction for industrial and consumptional use. The seepage often contains relatively high levels of bicarbonate, carbon dioxide and iron (Roelofs, 1983; Smolders and Roelofs, 1993). The combination of both high bicarbonate and carbon dioxide levels suppresses the pH (Stumm and Morgan, 1981) and thus the decomposition rate and the release of nutrients from the sediment to the water layer (Kok and Van de Laar, 1991). In some cases the external eutrophication and the decrease of seepage (Schot *et al.*, 1988) have been forced back and compensated for by the inlet of water with a low phosphate concentration (after artificial removal of phosphorus), for example in the shallow Loosdrecht Lakes (Van Liere *et al.*, 1989) and the Naardermeer (Barendregt *et al.* 1989), without leading to a improvement of the water quality till 1989.

Since the removal of phosphorus did not lead to direct improvement of the water quality and since we suppose that internal eutrophication is partly regulated by the macro-ions bicarbonate and sulphate (Roelofs, 1991), we have investigated the effects of alkalinity, acidity and external sulphate and phosphorus load on eutrophication processes in the water layer, by means of enclosure experiments (Bellemakers *et al.*, 1994). In the enclosures physical parameters (*e.g.* turbidity), chemical parameters (*e.g.* alkalinity, sulphate and ortho-phosphate) and biological parameters (*e.g.* dry weight of sessile algae) have been studied, in order to gain some insight in the eutrophication processes involved and to develop control measures for the improvement of the water quality in shallow lakes.

Study Site

The field experiments have been set up in the Naardermeer Nature Reserve (52° 17 N; 5° 8 W). The studied lake has a surface area of about 85 ha and an average depth of 0.8 m; it is surrounded by reed swamps, marshes and wood stands (Schot *et al.*, 1988). For a long time, the water of the lake was mesotrophic to eutrophic with a well-developed vegetation of floating and submerged macrophytes (van Leeuwen, 1977; van der Hoorn, 1980), such as *Ceratophyllum demersum* L., *Myriophyllum spicatum* L., *Najas marina* L., *Stratiotes aloides* L. and *Chara* spp., indicating alkalinities from 1 to 1.5 meq l^{-1} (De Lyon and Roelofs, 1986).

Gradual reduction of the amount of seepage by water extraction from the sandy areas east of the lake caused changes in the water chemistry and water quality. Since 1975 this water loss was compensated for by inlet of highly eutrophicated water from the river Vecht. After the inlet of this water, water quality decreased further; the turbidity increased and there was a deterioration of the development of submerged macrophyte stands. Since 1985 phosphate has been stripped from the inlet water. Yearly 1.1 x 10^6 m^3 water has been used, from which phosphate has been removed. Unfortunately, this measure did not cause improvement of water quality (Barendregt *et al.*, 1989).

Materials and Methods

Experimental design

In order to study the impact of the removal of phosphorus on the water quality of the Naardermeer, water samples of the river Vecht (before and after the removal of phosphorus; 1988) and of the Naardermeer (1988 and 1989) were monthly taken and analyzed in the laboratory. Ground water was sampled in and around the Naardermeer at 50 m below N.A.P. (Dutch standard level; n=10).

To measure the effects of manipulation on water chemistry transparent polycarbonate enclosures (height 1.5 m; Ø 1 m) were placed into the sediment, from August to December 1988 and from May to November 1989. The water inside the enclosures was isolated from the surrounding water by locating the enclosures 15 cm into the sediment. The water in the enclosures was treated as shown in Table 1. To reduce the alkalinity of the water in the enclosures to 0.5 meq l^{-1} diluted HCl was added. To study the effects of the removal of phosphorus, 100 l of water, collected just behind the phosphate removal installation, was added to the enclosures. To obtain the same flush in the control enclosures, water from outside the enclosures was used. This treatment was repeated every two weeks during the first two months of the experiment. Water samples were taken every two weeks from August 1988 to December 1988 and analyzed in the laboratory.

Table 1 Overview of the treatments in the enclosures

Exp 1	Exp 2	Treatments
1988	1989	Open water
1988	1989	Control
1988		Lowering of alkalinity
1988		Removal of phosphorus
1988		Lowering of alkalinity and removal of phosphorus
	1989	Precipitation of sulphate
	1989	Lowering of alkalinity and precipitation of sulphate

In 1989 an equivalent amount of barium chloride has been added to some enclosures, to study the effect of sulphate precipitation of the sediment Water samples were taken every two weeks from April 1989 to November 1989 and analyzed in the laboratory

The sessile algae were quantitatively removed from the inside and outside of the enclosures The sessile algae on the outer walls of the enclosures were sampled to obtain a measurement of their biomass in the open water environment

Field and laboratory measurements

Measurements of pH, alkalinity and turbidity were carried out within a few hours after sampling pH was measured with a GK2501B combined pH electrode, connected to a Radiometer Copenhagen PHM82 pH/mV meter Alkalinity was determined by titrating 100 ml of water with 0 01 N HCl down to pH 4 2 (Stumm and Morgan, 1981) A part of each sample (50 ml) was passed through a Whatman GF/C filter (1 2 µm) These samples were stored in iodated polyethylene bottles at -28 °C until chemical analysis Sulphate concentrations were determined gravimetrically, according to Technicon Auto-analyzer Methodology (1981) The filters with the accumulated seston were stored in aluminum foil at -28 °C until chlorophyll-a analysis, according to Roijackers (1981) The dry weight of sessile algae has been measured by drying at 105 °C of 1 dm^2 scrapings

All data were transformed and geometric means and the 95% confidence limits were calculated The significances of differences in water chemistry in the various enclosures and the sampled locations during the sampling period have been tested with the paired T-test and differences in the period have been tested by using the T-test (Sokal and Rohlf, 1981)

Results

After the removal of phosphorus from the inlet water the ortho-phosphate concentration, the alkalinity, chlorophyll-a and turbidity were decreased significantly (Table 2) However, in the water of the lake chlorophyll-a and turbidity were much higher than in the inlet water, whereas the ortho-phosphate concentration remained low The sulphate concentrations in the inlet water (before and after removal

Table 2: Geometric means and 95% confidence limits of the chemical composition of the ground water, inlet water and Naardermeer water during the period July 1988 - December 1988 and July 1989 - December 1989. Different letters (a, b or c) indicate statistical differences between the four sampled locations at the 5% level, according to the paired T-test ($p < 0.05$). The asterisks indicate statistical difference between 1988 and 1989, according to the T-test; *: $p < 0.05$ and **: $p < 0.01$.

		Ground water (1986)		Inlet water before removal of phosphorus		Inlet water after removal of phosphorus		Naardermeer (1988)		Naardermeer (1989)	
pH		7.2	(6.4-7.9)	7.7	(n=1)	6.5	(n=1)	8.0	(7.6-8.4)	8.0	(7.8-8.1)
Alkalinity	(meq l^{-1})	1.6	(1.1-2.1)	1.7	(1.6-1.9)^a	0.9	(0.7-1.8)^b	1.9	(1.8-2.0)^a	1.8	(1.7-2.0)
Sulphate	(mmol l^{-1})	0.15	(0.02-0.29)	1.54	(1.48-1.60)^b	1.46	(1.36-1.56)^b	0.87	(0.79-0.97)^c	0.96	(0.80-1.15)
o-Phosphate	(μmol l^{-1})	0.05	(0.02-0.07)	0.40	(0.23-0.69)^b	0.06	(0.05-0.08)^a	0.09	(0.05-0.16)^a	0.17	(0.11-0.26)*
Chlorophyll-a	(μg l^{-1})	-		25	(16-38)^a	8	(4-18)^b	62	(52-74)^c	10	(6-17)**
Turbidity	(ppm $PtCl_2$)	-		9	(6-13)^a	3	(2-4)^b	14	(10-20)^a	4	(2-7)**
number of observations		10		5		5		12		12	

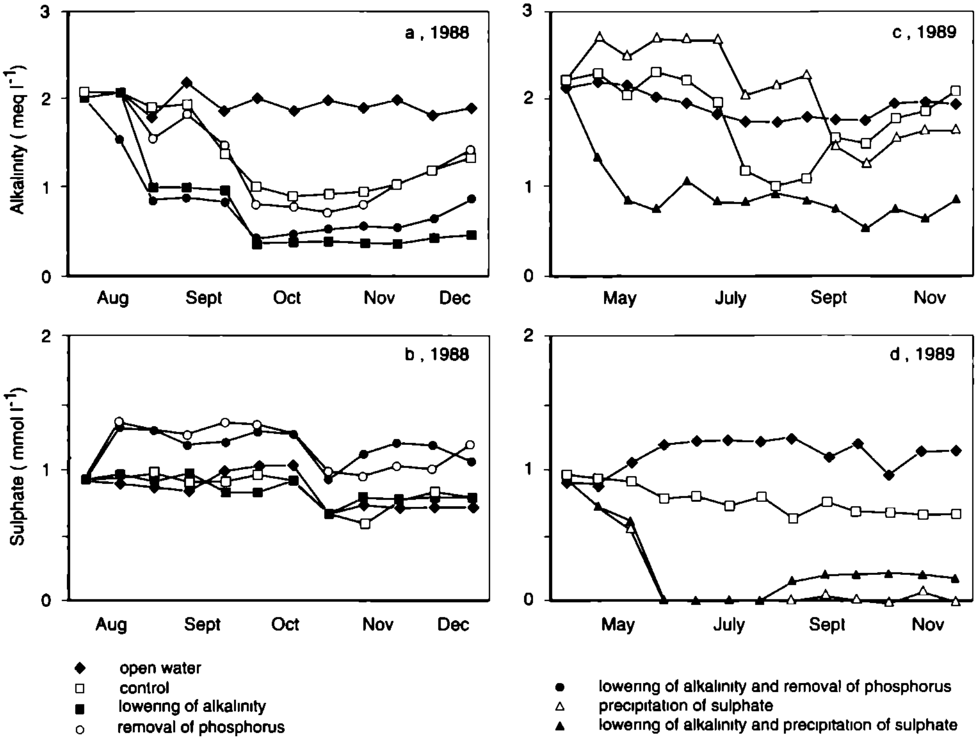


Figure 1: Changes of alkalinity and sulphate in the enclosures and the open water of the Naardermeer.

of phosphorus) were significantly higher than in the water of the lake. The sulphate and ortho-phosphate concentration of the ground water were significantly lower (0.15 mmol l^{-1} and 0.05 μmol l^{-1} respectively; Table 2), compared with the water of the River Vecht (1.54 mmol l^{-1} and 0.40 μmol l^{-1} respectively) before the removal of phosphorus. pH and alkalinity were about the same in the ground water, in the inlet water before removal of phosphorus and in the Naardermeer.

After initiating the first set of experiments (August 1988) there was a significant decrease of alkalinity (to 0.9 meq l^{-1}) observed in the enclosures without artificial decreases in alkalinity (Figure 1[a]), but this decrease was not as large as in the enclosures with artificial decreases in alkalinity. The alkalinity of the open water remained at the same level (ca 2 meq l^{-1}) during the experimental period.

After adding pre-cleaned Vecht water, poor in phosphates, to the enclosures of the first set of experiments (1988), there was a significant increase ($p < 0.01$) in sulphate content (Figure 1[b]). The p-values for all statistical differences of the enclosure experiments are presented in the Appendix.

After two months (October) turbidity and chlorophyll-a in the enclosures without artificial decreases in alkalinity had reached maximum values of 20-30 ppm $PtCl_2$ and 60-100 μg l^{-1} respectively (Figure 2[a, b]). At the same time, the values of the open water remained somewhat lower (about 10 ppm $PtCl_2$ and 40 μg l^{-1} respectively), compared with those of the control enclosures. In the enclosures

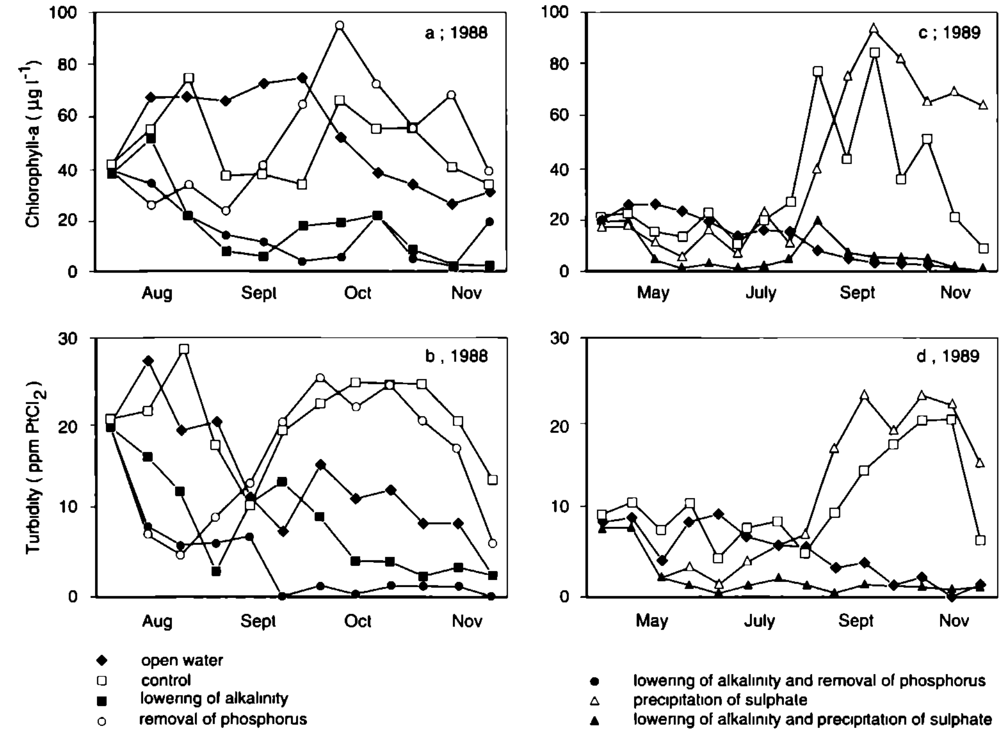


Figure 2: Changes of chlorophyll-a and turbidity in the enclosures and the open water of the Naardermeer.

with artificial decreases in alkalinity the turbidity and the chlorophyll-a content decreased to very low levels.

At the end of these experiments (November) turbidity and chlorophyll-a content decreased, in all enclosures. In 1989, compared to 1988, a slight increase of sulphate and significant decreases of turbidity and chlorophyll-a content were measured in the lake water (Table 2).

In the second set of experiments (1989) the effects of a barium-treatment with or without a decrease of alkalinity has been studied. Addition of barium led to barium levels in the water layer of 60

Table 3: Biomass of sessile algae (expressed in mg dry weight cm^{-2}) on artificial substrates.

Treatment	Dry weight
Open water	273
Control	100
Lowering of alkalinity	37
Removal of phosphorus	118
Lowering of alkalinity and removal of phosphorus	22
Precipitation of sulphate	458
Lowering of alkalinity and precipitation of sulphate	172

μmol l^{-1} and 314 μmol l^{-1} respectively, for the combined alkalinity/barium treatment and the barium treatment. The addition of barium only led to an increase of alkalinity of the water layer in the enclosure (up to 2.6 meq l^{-1}). This increase was significantly different from the control (Figure 1[c]; $p < 0.01$). The barium-treated enclosures in which alkalinity was also artificially reduced showed the lowest alkalinity values (0.8 meq l^{-1}).

Addition of barium to the water caused significant effects on turbidity and chlorophyll-a contents (Figure 2[c, d]). The samples of the open water and of the enclosures with barium decreased in alkalinity were clear (low turbidity and low levels of chlorophyll-a). In the barium and control enclosures, after four months maximum values, up to 22 ppm $PtCl_2$ for turbidity and 95 μg l^{-1} for chlorophyll-a were reached.

The amount of sessile algae was very low in the enclosures after lowering of alkalinity (Table 3). In enclosures treated with barium, the amount of sessile algae was highest.

Discussion

In order to compensate for the shortage of seepage to the Naardermeer, inlet of river Vecht water, rich in phosphate and sulphate has been applied. From 1985 onward, after the construction of a phosphate removal installation, the phosphate content of the inlet water has decreased considerably. Nevertheless, after 1985 the water quality of the lake did not improve the first years after inlet of water. During the period 1990 until 1992, there was an improvement of the water quality observed, probably due to a drought period. During this period, there was less water led in from the river Vecht to the Naardermeer.

The pH and bicarbonate plays an important role in the rate of decomposition of the organic sediment and thus in the internal eutrophication (Brock *et al.*, 1985; Kok and Van de Laar, 1991). It can be concluded from our experiments that the pH and alkalinity (bicarbonate concentration) of the water decreased after the removal of phosphorus, whereas in the open water of the Naardermeer both remained at levels, comparable with the levels before the removal of phosphorus. This can be explained by the method of removal of phosphorus: the water of the river Vecht has been led through $FeCl_3$, to precipitate $FePO_4$·HCl is also added to keep $FeCl_3$ soluble, which causes the decrease of pH and alkalinity (Stumm and Morgan, 1981). The return to higher levels of pH and alkalinity can be explained by the previously described internal alkalinisation by means of reductive processes *e.g.* sulphate reduction, which generates in-lake alkalinity (Boström *et al.*, 1982; Schindler, 1986; Van Dam, 1988) and stimulates mineralisation and phosphorus mobilisation (Mortimer, 1971). Removal of phosphorus from the inlet water appeared not favourable for the water quality in the Naardermeer. However, the sulphate concentration remained high in the inlet water, despite of the removal of phosphorus, whereas in the ground water, that in the past seeped in, it is still very low. The sulphate concentration has been even doubled in the open water of the lake from 427 μmol l^{-1} in 1976 (Leentvaar, 1976) to 870 μmol l^{-1} and 960 μmol l^{-1} in the late 1980s (Table 2). This increase in sulphate may cause an increase of sulphate reduction in the sediment (Schuurkes *et al.*, 1988; Smolders and Roelofs, 1993), stimulating mineralisation and phosphorus mobilisation.

To study this hypotheses, we experimentally increased the flux of sulphate to the sediment by the addition of barium chloride. Because of the possible toxic effects of barium, the concentrations of this element were measured in the enclosures. These concentrations (60-314 μmol l^{-1}) were much lower than the concentrations Stanley (1974) found (300-820 μmol l^{-1}) to cause a 50% growth inhibition of the macrophyte *Myriophyllum spicatum* L.

The alkalinity significantly increased after this treatment (Figure 1^c) and the biomass of the sessile algae was enhanced, too (Table 3). This indicates that phosphate release from the sediment increased after the sulphate treatment, and replenished the phosphate removed from the inlet water.

Canfield (1983) and Curtis (1989) also found an enhanced availability of phosphate as a result of the sulphate reduction in alkaline environments. Furthermore, it was observed that the artificial decrease in alkalinity led to an improvement of the water quality. Enhanced sulphate fluxes to sediments may play a crucial role in the internal eutrophication processes in lakes.

According to the hypothesis of Phillips *et al.* (1978) macrophytes decline in eutrophicated lakes where the cycle of nutrients is disturbed. Introduction of alkaline, sulphate rich water into the Naardermeer disturbed the cycle of nutrients. The release of nutrients from the sediment leads to high chlorophyll-a contents and turbidity of the water. Since light is an important factor for germination and development of submerged macrophytes (Smits *et al.*, 1990), the present circumstances are not likely to change after inlet of water with low phosphate concentrations.

From the results of these enclosure experiments it is recommended that both phosphorus and sulphate concentrations in the Naardermeer should be reduced. The best option, however, is to restore the former hydrological conditions, in which the system is fed by ground water relatively rich in bicarbonate and carbon dioxide and poor in sulphate.

Acknowledgements

This study has been financed by the Ministry of Housing, Physical Planning and Environment, Directorate Water (Project 011661). We are grateful to Dr. R. Bobbink, Dr. A. Smolders, Dr. J.G.M. Roelofs, Prof. Dr. C. den Hartog and Dr. H. van Dam for critically reading this manuscript. We also thank the following students: M.W.A. Melis, J. Stapel, T.M. Tahey and P.J.A. Vervuren for practical assistance in the field and experimental work. The ground water data have been kindly provided by Dr. A. Barendregt, Dept. of Environmental Science of the University of Utrecht.

Appendix A

Statistics of the chemical data of the enclosure experiments (1988): p-values of the paired T-Test. I: control; II: removal of phosphorus; III: lowering of alkalinity; IV: lowering of alkalinity and removal of phosphorus and V: Open water. n.s.: not significant.

	I-II	I-III	I-IV	I-V	II-III
Alkalinity	n.s.	<0.01	<0.05	<0.01	<0.01
Sulphate	<0.01	<0.01	n.s.	n.s.	n.s.
Chlorophyll-a	n.s	<0.01	<0.01	n.s.	<0.01
Turbidity	<0.05	<0.01	<0.01	<0.01	<0.01

	II-IV	II-V	III-IV	III-V	IV-V
Alkalinity	<0.05	<0.01	n.s.	<0.01	<0.01
Sulphate	<0.01	<0.01	<0.01	<0.01	n.s.
Chlorophyll-a	<0.01	n.s.	n.s.	<0.01	<0.01
Turbidity	<0.05	n.s.	<0.01	<0.01	n.s.

Statistics of the chemical data of the enclosure experiments (1989): p-values of the paired T-Test. I: control; II: precipitation of sulphate; III: lowering of alkalinity; precipitation of sulphate and IV: Open water. n.s.: not significant.

	I-II	I-III	I-IV	II-III	II-IV	III-IV
Alkalinity	n.s.	<0.01	n.s.	<0.01	n.s.	<0.01
Sulphate	<0.01	<0.01	<0.05	n.s.	<0.01	<0.01
Chlorophyll-a	n.s.	<0.01	<0.05	<0.01	<0.05	<0.05
Turbidity	n.s.	<0.01	<0.01	<0.01	n.s.	<0.05

References

Barendregt, A , Wassen, M J , Schot, PP, Aaldennk, R H , Verstraelen, PJ T and Straathof, N , 1989 De suppletie van het Naardermeer in relatie tot natuurbeheer In L van Liere, R M M Roijackers and PJ T Verstraelen (eds), Integraal waterbeheer in het Goois/Utrechts stuwwallen- en plassengebied, Elsevier, Amsterdam, p 196-212 (in Dutch)

Bellemakers, M J S , Maessen, M and Roelofs, J G M , 1994 Effects of liming on water chemistry in shallow acidified pools in the Netherlands enclosure experiments Water, Air, Soil Pollut 73 131-142

Bostrom, B , Jansson, M and Forsberg, C , 1982 Phosphorus release from lake sediments Arch Hydrobiol Beih /Ergebn Limnol 18 5-60

Brock, Th C M , Boon, J J and Paffen, B G P, 1985 The effects of the season and water chemistry on the decomposition of *Nymphaea alba* L , weight loss and pyrolysis mass spectrometry of the particulate matter Aquat Bot 22 197-229

Canfield, D E , 1983 Sensitivity of Florida lakes to acid precipitation Water Resourc Res 19 833-846

Curtis, PJ , 1989 Effects of hydrogen ion and sulphate on the phosphorus cycle of a Precambrian Shield lake Nature 337 156-158

De Lyon, M and Roelofs, J G M , 1986 Waterplanten in relatie tot waterkwaliteit en bodemgesteldheid Parts 1 and 2 Laboratory of Aquatic Ecology, University of of Nijmegen, 232 pp (in Dutch)

Forsberg, C , 1989 Importance of sediments in understanding nutrient cycling in lakes Hydrobiologia 176/177 263-277

Koerselman, W and Verhoeven, J T A , 1989 Effecten van infiltratie van gebiedsvreemd water op trilvenen in het Noorderpark In J G M Roelofs (eds), Aanvoer van gebiedsvreemd water omvang en effecten op oecosystemen Laboratory of Aquatic Ecology and Biogeology, University of Nijmegen, p 32-51 (in Dutch)

Kok, C J and Van de Laar, B J , 1991 Influence of pH and buffering capacity on the decomposition of *Nymphaea alba* L detritus in laboratory experiments a possible explanation for the inhibition of decomposition at low alkalinity Verh Internat Verein Limnol 24 2689-2692

Leentvaar, P, 1976 Hydrobiologie van het Naardermeer PA Bakker, C A J van der Hoeven-Loos, L R Mur en A Stork (eds) Published by Stichting Commissie voor de Vecht en het Oostelijk en Westelijk Plassengebied De Noordelijke Vechtplassen, 393 pp (in Dutch)

Mortimer, C H , 1971 Chemical exchanges between sediments and water in the Great Lakes, speculations on probable regulatory mechanisms Limnol Oceanogr 16 387-404

Phillips, G L , Eminson, H and Moss, B , 1978 A mechanism to account for macrophyte decline in progressively eutrophicated freshwaters Aquat Bot 4 103-126

Roelofs, J G M , 1983 Impact of acidification and eutrophication on macrophyte communities in soft waters in the Netherlands I Field observations Aquat Bot 17 139-155

Roelofs, J G M , 1991 Inlet of alkaline river water into peaty lowlands effects on water quality and *Stratiotes aloides* L stands Aquat Bot 39 267-293

Roijackers, R , 1981 A comparison between two methods of extracting chlorophyll-a from different phytoplankton samples Hydrobiol Bull 15 179-183

Sand-Jensen, K. and Søndergaard, M., 1981. Phytoplankton and epiphyte development and their shading effect on submerged macrophytes in lakes of different nutrient status. Int. Rev. ges. Hydrobiol. 66: 529-552.

Schindler, D.W., 1986. The significance of in-lake production of alkalinity. Water, Air, Soil Pollut. 30: 931-944.

Schot, P.P., Barendregt, A. and Wassen, M.J., 1988. Hydrology of the wetland Naardermeer: influence of the surrounding area and impact on vegetation. In: J.W. Van Hoorn (Ed.): Agrohydrology - recent development. Elsevier, Amsterdam. 550 pp. / Agriculture Water Management 14: 459-470.

Schuurkes, J.A.A.R., Kempers A.J. and Kok, C.J., 1988. Aspects of biochemical sulphur conversions in sediments of a shallow soft water lake. J. Freshw. Ecol. 4: 369-381.

Smits, A.J.M., Van Avesaath, P.H. and Van der Velde, G., 1990. Germination requirements and seed banks of some nymphaeid macrophytes (*Nymphaea alba* L., *Nuphar lutea* (L.) Sm. and *Nymphoides peltata* (Gmel.) O. Kuntze). Freshwat. Biol. 24: 315-326.

Smolders, A. and Roelofs, J.G.M., 1993. Sulphate-mediated iron limitation and eutrophication in aquatic ecosystems. Aquat. Bot. 46: 247-253.

Sokal, R.R. and Rohlf, F.J., 1981. Biometry, second edition. Freeman, San Francisco, 859 pp.

Stanley, R.A., 1974. Toxicity of heavy metals and salts to Eurasian watermilfoil (*Myriophyllum spicatum* L.). Arch. Environ. Contam. Toxicol. 2: 331-341.

Stumm, W. and Morgan, J.J., 1981. Aquatic Chemistry. Wiley and Sons, 780 pp.

Technicon Auto-Analyzer Methodology, 1981. Industrial Method 635-81W, New York.

Van Dam, H., 1988. Acidification of three moorland pools in the Netherlands by acid precipitation and extreme drought periods. Freshwat. Biol. 20 : 157-176.

Van den Hoorn, D.A.C., 1980. Het Naardermeer, gisteren, vandaag en morgen. Vereniging tot Behoud van Natuurmonumenten in Nederland. Natuurmonumenten, 's Graveland, 47 pp. (in Dutch).

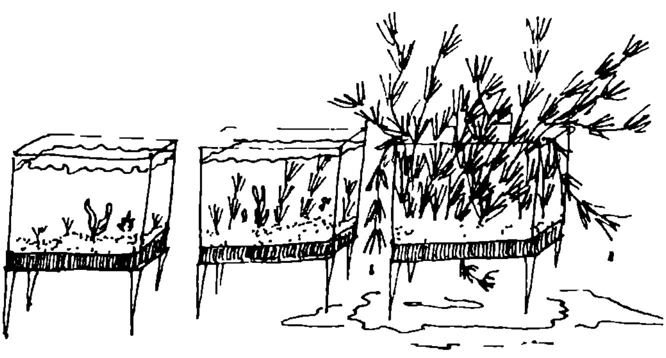
Van Leeuwen, B.L.J., 1977. Naardermeer. In: A. Coops, F.W. Maas and A. Scheygrond (eds.), Natuurmonumenten in Nederland, Uitgeverij Luitingh B.V., Laren N.H., p. 69-71 (in Dutch).

Van Liere, L., Breebaart, L., Kats, W. and Buysse, J.J., 1989. De waterkwaliteit in het Loosdrechtse Plassengebied. In: L. van Liere, R.M.M. Royackers and P.J.T. Verstraelen (eds.), Integraal waterbeheer in het Goois/Utrechts Stuwwallen- en Plassengebied, Elsevier, Amsterdam, p. 265-278 (in Dutch).

4

Effects of Liming on Shallow Acidified Moorland Pools

a Culture and a Seed Bank Experiment



Abstract

To study the effects of liming on water chemistry and the development of aquatic macrophytes, culture experiments with *Littorella uniflora* (L.) Aschers. and *Juncus bulbosus* L. were carried out. The addition of powdered limestone to acidified moorland pools lead to increased phosphate levels in the water layer and increased carbon dioxide levels in the sediment pore water and the water column; this resulted in an excessive growth of *Juncus bulbosus*. Similar observations were made in seed bank experiments, conducted in a greenhouse. After 2 years the abundancies of *J. bulbosus* were very high. *Littorella uniflora* and *Lobelia dortmanna* L. germinated spontaneously and developed in mini-ecosystems with sediments from the Padvindersven and the Peetersven, respectively, two moorland pools from where these species had disappeared a few decades ago. Spontaneous germination of and development of endangered soft water macrophytes only occurred on sediments after removal of the organic sapropelium layer. Therefore, this study showed that in order to restore the original water chemistry of acidified moorland pools, removal of the organic sapropelium layer is necessary, as well for improving the vegetation development.

M.J.S. Bellemakers, M. Maessen, G.M. Verheggen and J.G.M. Roelofs, 1996. Effects of liming on shallow acidified moorland pools: a culture and a seed bank experiment. Aquatic Botany 54: 37-50.

Introduction

In recent decades, the adverse effects of acid precipitation on rivers and lakes have been observed in Canada, the USA and Scandinavia (Drabløs and Tollan, 1980). Meanwhile, it has been demonstrated that acidification alters the water chemistry and biota of rivers and lakes (Henriksen, 1980; Muniz, 1991; Schindler *et al.*, 1991). Atmospheric deposition also had a severe impact on the water chemistry and the biodiversity of moorland pools in the Netherlands (Van Dam *et al.*, 1981; Roelofs, 1983; Leuven, 1988). These shallow pools were formerly characterized by several representatives of the phytosociological alliance Littorellion (Schoof - Van Pelt, 1973), which is characteristic of oligotrophic, weakly buffered shallow waters (Westhoff and Den Held, 1973). Species belonging to the Littorellion communities are *Littorella uniflora* (L.) Aschers., *Lobelia dortmanna* L., *Echinodorus repens* (Lamk.) Kern et Reichgelt, *Isoetes lacustris* L., *Isoetes echinospora* Durieu and *Luronium natans* (L.) Rafin. Acidification seriously threatens the communities of the Littorellion (Van Dam *et al.*, 1981; Roelofs, 1983; Arts, 1990).

Measures to counteract the negative effects of acidification are necessary, because acidic deposition is still high, in spite of all efforts to reduce it (Den Hartog, 1993). Hultberg and Anderson (1982) and Baalsrud *et al.* (1985) have carried out liming experiments and concluded that liming must be planned and adapted to local conditions in order to ensure good results. The impact of liming on acidified lakes in the USA has been evaluated by Schofield *et al.* (1986) and Brown and Goodyear (1987). They concluded that simple whole-lake liming practices would not be adequate for maintaining the desired water quality, because of the complexity of water chemistry and water sediment interactions.

In earlier studies, the effects of liming on water chemistry and diatoms in shallow moorland pools were studied with enclosures (Van Dam *et al.*, 1988; Bellemakers *et al.*, 1994). As variability at the ecosystem level had influenced the outcome of these enclosure experiments, culture and seed bank experiments with macrophytes from the communities of the Littorellion have been set up under more constant conditions to determine the effects of liming on shallow weakly buffered moorland pools and the spontaneous germination and development of endangered soft water macrophytes. In these experiments all factors were kept constant, with the exception of pH, alkalinity and removal of the organic sapropelium layer, which were the variables.

During the process of acidification, an increased dominance of *Juncus bulbosus* L. has been observed after an increase of carbon dioxide, an increase of ammonium and a decrease of nitrate in the sediment pore water (Roelofs, 1983). Earlier experiments showed that after liming in field situations internal eutrophication could take place (Roelofs, 1991; Bellemakers *et al.*, 1994; Roelofs *et al.*, 1994). It is, therefore, important to find out whether liming of the water layer also causes changes in the carbon and nitrogen budget leading to massive expansion of *Juncus bulbosus* These experiments can also give indications, whether the sediment type (mineral of organic) also influences the massive expansion of *Juncus bulbosus*

Materials and Methods

The culture experiment with transplants

This experiment was conducted from September 1988 until January 1989 Plants of *Littorella uniflora* and *Juncus bulbosus* were collected from the Beuven and the Ven bij Schaijk, respectively The plants were placed in plastic pots (225 ml), filled with mineral or organic sediment The mineral sediment was collected from the Voorste Goorven and the organic sediment from the Ven bij Schaijk (Figure 1)

The culture experiment was carried out in duplicate in a temperature-controlled water bath (20° C) with artificial light (HP1, 16 hrs, 200 µE m^{-2} s^{-1} at the water surface) for a period of 4 months All pots were placed in glass aquaria (0 3 m x 0 2 m x 0 2 m) with the culture medium containing 50 mg sea salt, 17 mg $CaSO_4$ H_2O and 0 01 mg KNO_3 l^{-1} demineralised water The effects of alkalinity were studied in a gradient of HCO_3^- (0 0, 0 1, 0 2, 0 4, 1 0 and 2 0 meq l^{-1} HCO_3^-), on both mineral and organic sediments The level of the alkalinity was maintained by adjustment with equivalent amounts of $NaHCO_3$ and $CaCl_2$ solutions

The pH and alkalinity were measured weekly and water samples were collected monthly in polyethylene bottles during the whole experimental period After filtration the water samples were stored at -28 °C prior to chemical analysis

The biomass of the submerged macrophytes was determined after cleaning and drying the fresh plant material for 24 hrs at 105 °C at the end of the experiment The change in biomass percentage was calculated as

$$\Delta \text{ biomass} = \frac{B_{\text{final}} - B_{\text{initial}}}{B_{\text{initial}}} \times 100\%$$

In this formula, B_{initial} is the biomass (dry weight) at the beginning of the experiment and the B_{final} the biomass at the end of the experiment

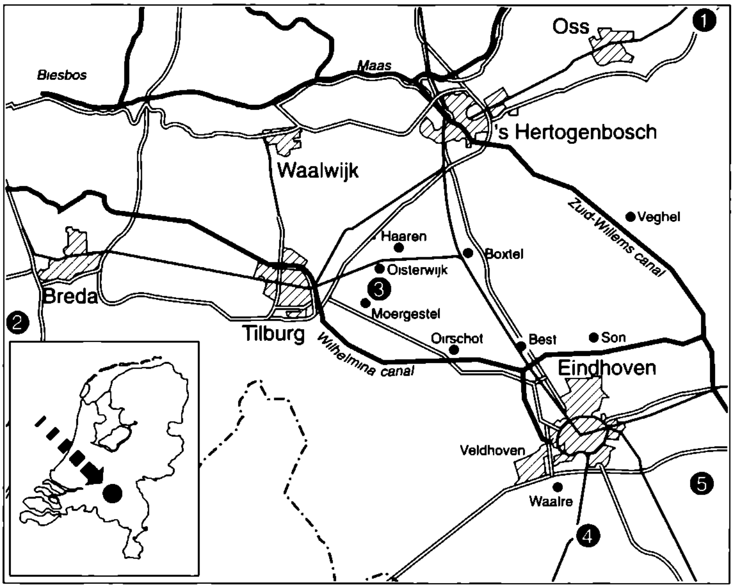


Figure 1: Location of the studied moorland pools: (1) Ven bij Schaijk; (2) Padvindersven; (3) Voorste Goorven; (4) Peetersven; (5) Beuven.

The seed bank experiment

To study the effects of a reduction of the atmospheric deposition and the effects of increased pH and alkalinity on the germination of seeds of submerged macrophytes and helophytes, sediments of several acidified (Ven bij Schaijk, Padvindersven and Voorste Goorven) or eutrophicated (Beuven and Peetersven) (Figure 1) moorland pools were collected. The experiment was conducted in a greenhouse.

This experiment was carried out in two stages. The artificial rain experiment, simulating reduced atmospheric deposition, was carried out from May 1985 until October 1985 and the liming experiment from November 1988 until September 1990.

Ten small-scale aquatic systems (mini-ecosystems) were created using black PET containers (0.8 m x 0.8 m x 0.3 m). These containers were filled with sediments of the above-mentioned moorland pools. To study the effects of a reduction in atmospheric deposition, the mini-ecosystems with Beuven and Peetersven sediments were exposed to two different rain treatments: clean (0 mmol l^{-1} $(NH_4)_2SO_4$) and polluted (5 mmol l^{-1} $(NH_4)_2SO_4$) rain water. To compensate for evaporation, culture medium, containing 1.18 mg $Ca(NO_3)_2 \cdot 4H_2O$, 0.14 mg KH_2PO_4, 0.44 mg K_2SO_4, 3.05 mg $MgCl_2 \cdot 6H_2O$, 1.17 mg NaCl and 2.55 mg $NaNO_3$ l^{-1} demineralised water, was added. The water layers of the other mini-ecosystems were limed with limestone (powdered marlstone; 2 meq l^{-1} = 0.1 g l^{-1}). The buffering was performed by adding equivalent amounts of $NaHCO_3$ and $CaCl_2$ solutions up to an alkalinity of 0.2 meq l^{-1}.

Water samples were taken monthly and subsequently analysed. The germination and the development of (submerged) macrophytes were recorded qualitatively. The abundance per species was estimated in percent coverage at the end of the experiment.

Laboratory measurements

The water content of the fresh sediment was measured by weighing (10 g for 24 hrs at 105 °C) and reweighing. Water-soluble ions were extracted from 60 g fresh sediment by continuous agitation for 1 h (100 rev min^{-1}) in 180 ml distilled water. The water extract of each sediment sample was then centrifuged for 10 min at 11000 rev min^{-1} and stored at -28 °C. The organic content of the sediment was expressed as the loss on ignition at 550 °C during 4 hrs.

pH and alkalinity were measured within 2 hours of sampling: pH with a GK2501B pH electrode connected to a Radiometer Copenhagen PHM82 pH/mV meter, and alkalinity by titration of a subsample with 0.01 N HCl down to pH 4.2. Acidity was estimated by titration of a subsample with 0.01 N NaOH up to pH 8.2. Carbon dioxide was calculated from acidity and bicarbonate from alkalinity, both after correction for pH. Water samples for analysis of nutrients were filtered through a Whatman GF/C filter (1.2 µm) and stored in iodated polyethylene bottles at -28 °C.

Chemical analysis were done with a Technicon Auto-analyzer (Technicon Auto-analyzer Methodology, 1981). Sulphate was measured gravimetrically, using barium chloride. Colorimetric determinations were used for ortho-phosphate (Henriksen, 1965), ammonium (Grasshoff and Johannsen, 1977), nitrate and nitrite (Kamphake *et al.*, 1967) and chloride (O'Brien, 1962). Calcium, magnesium, aluminum and total-phosphate were determined by the Inductive Coupled Plasma Method (Instrument Laboratory Plasma 2000).

Statistics

All data were computed according to SAS (1990[a]). The water chemistry data from the different treatments in the culture experiment were tested statistically using the signed-rank Wilcoxon test. The Spearman rank correlation test was used to analyse correlations between the different variables. The differences between nutrient concentrations and increase of biomass of submerged macrophytes as a result of different alkalinities, were tested statistically using the ANOVA multiple comparison between all means (Sokal and Rohlf, 1981; SAS, 1990[b]).

Table 1: The chemical characteristics (means) of the water layer and the water extracts of the sediment of the examined moorland pools (n=3). HCO_3^- is expressed in meq l^{-1}, loss on ignition in percent and all nutrients and metal concentrations in µmol l^{-1} (water layer) or µmol kg^{-1} (sediments). VGV, Voorste Goorven (mineral sediment), VS, Ven bij Schaijk (organic sediment), PV, Padvindersven (organic sediment), BV, Beuven (mineral sediment) and PeV, Peetersven (mineral sediment).

	mineral sediment			organic sediment	
	BV	VGV	PeV	VS	PV
Water					
pH	5.9	4.8	4.1	4.4	4.6
HCO_3	0.1	0.1	0.0	0.0	0.0
Ammonium	20	130	20	27	11
Nitrate	36.0	13.2	8.0	1.0	5.7
Orthophosphate	0.81	0.27	0.28	0.18	0.44
Aluminum	6	11	7	46	11
Calcium	383	61	375	164	101
Magnesium	134	57	135	177	56
Sulphate	519	223	279	521	261
Sediment					
pH	5.6	5.6	5.9	5.7	5.1
HCO_3	0.1	0.0	0.1	0.0	0.1
% moisture	13.0	21.3	36.0	68.1	70.0
Loss on ignition	1.1	1.5	7.9	8.4	8.4
Ammonium	540	769	51	500	500
Nitrate	87	65	5	112	112
Orthophosphate	8.3	1.8	1.0	10.1	10.1
Aluminum	98	187	9	11	11
Calcium	590	1080	138	492	492
Magnesium	186	367	39	719	63
Sulphate	991	3454	188	3425	925

Chemical characteristics of the study sites

All sites are situated in the southeastern part of the Netherlands (Figure 1). All of the pools investigated (Ven bij Schaijk, Padvindersven, Voorste Goorven and Peetersven) have been acidified in recent decades, with the exception of the Beuven. This shallow lake was enriched by inlet of enriched brook water and later threatened by acidification after removal of the sapropelium layer (Buskens, 1989).

The chemical characteristics are presented in Table 1. The water layers of the Voorste Goorven, the Ven bij Schaijk, the Padvindersven and the Peetersven are very acid. The pH of the weakly buffered Beuven is relatively high (5.9). Calcium and magnesium concentrations are low in the water layer of these moorland pools (Table 1), except for the Beuven and the Peetersven.

Table 2: The chemical composition (geometric means and 95% confidence limits) of the water layer of the different treatments during the culture experiment. HCO_3 and CO_2 (n=26) are expressed in meq l^{-1} and all nutrients and metal concentrations (n=9) are expressed in µmol l^{-1}. Min.=mineral, org.=organic and alk.=alkalinity.

	min. alk.=0		min. alk.=0.1		min. alk.=0.2		min. alk.=0.4		min. alk.=1.0		min. alk.=2.0	
pH (n=26)	4.6	(4.2-5.1)	6.5	(6.2-6.7)	7.3	(7.1-7.5)	7.7	(7.4-8.0)	8.4	(8.2-8.5)	8.6	(8.4-8.8)
HCO_3	0.0	(0.0-0.0)	0.1	(0.1-0.1)	0.2	(0.2-0.2)	0.4	(0.4-0.4)	1.0	(0.8-1.1)	2.0	(1.7-2.3)
CO_2	missing value		0.08	(0.06-0.10)	0.03	(0.03-0.03)	0.02	(0.02-0.02)	0.01	(0.01-0.01)	0.01	(0.01-0.01)
Calcium	185	(126-270)	162	(119-220)	158	(122-205)	130	(103-164)	121	(107-137)	92	(87-98)
Magnesium	101	(71-147)	97	(71-134)	93	(68-128)	85	(62-115)	79	(61-102)	60	(48-75)
Sulphate	257	(154-429)	268	(153-470)	269	(151-481)	259	(146-463)	285	(165-495)	278	(153-504)
o-phosphate	0.13	(0.01-0.23)	0.17	(0.06-0.30)	0.19	(0.11-0.33)	0.21	(0.12-0.37)	0.27	(0.19-0.39)	0.31	(0.16-0.60)
Ammonium	3.5	(1.9-6.2)	2.6	(1.5-4.2)	3.1	(1.9-5.1)	2.9	(1.5-5.8)	3.6	(1.9-6.8)	3.3	(1.7-6.6)
Nitrate	0.9	(0.6-1.4)	0.6	(0.3-0.9)	0.7	(0.5-1.1)	0.9	(0.6-1.3)	0.8	(0.6-1.2)	0.9	(0.7-1.2)
Chloride	862	(661-1123)	796	(622-1019)	824	(642-1057)	760	(581-993)	791	(594-1055)	768	(614-961)
	org. alk.=0		**org. alk.=0.1**		**org. alk.=0.2**		**org. alk.=0.4**		**org. alk.=1.0**		**org. alk.=2.0**	
pH (n=26)	4.8	(4.4-5.3)	6.7	(6.5-6.9)	7.1	(6.9-7.3)	7.7	(7.3-8.1)	8.0	(7.8-8.2)	8.4	(8.1-8.6)
HCO_3	0.0	(0.0-0.0)	0.1	(0.1-0.1)	0.2	(0.2-0.2)	0.4	(0.3-0.4)	1.0	(0.8-1.1)	2.0	(1.7-2.3)
CO_2	missing value		0.05	(0.03-0.05)	0.04	(0.03-0.05)	0.02	(0.02-0.02)	0.02	(0.02-0.02)	0.02	(0.02-0.02)
Calcium	162	(121-215)	143	(114-180)	130	(106-161)	121	(110-133)	103	(91-118)	79	(56-110)
Magnesium	137	(92-204)	128	(90-183)	119	(86-166)	105	(80-138)	90	(74-109)	65	(46-92)
Sulphate	251	(156-405)	269	(162-448)	266	(154-457)	269	(164-441)	281	(169-468)	297	(171-514)
o-phosphate	0.11	(0.07-0.16)	0.20	(0.11-0.38)	0.14	(0.10-0.20)	0.28	(0.19-0.41)	0.29	(0.18-0.48)	0.40	(0.23-0.68)
Ammonium	2.4	(1.3-4.1)	2.5	(1.9-3.4)	2.7	(1.7-4.4)	2.5	(1.5-4.3)	3.5	(2.3-5.2)	2.6	(1.8-3.7)
Nitrate	1.1	(0.4-3.4)	1.0	(0.4-2.2)	1.1	(0.4-2.9)	1.1	(0.5-2.5)	1.2	(0.6-2.4)	1.3	(0.7-2.3)
Chloride	841	(650-1088)	789	(615-1012)	821	(640-1053)	772	(601-993)	792	(620-1013)	809	(583-1123)

All sediments of these shallow lakes are sandy, in some cases covered by an organic sapropelium layer. Since the sediments are non-calcareous or slightly calcareous, the interstitial water of these sediments is weakly acid; pH ranges from 5.1 to 5.9 and alkalinity from 0.0 to 0.1 meq l^{-1} in the interstitial water. The ammonium, nitrate, aluminum and sulphate concentrations are high in the interstitial water, except for the Peetersven.

Results

Culture experiment

Water chemistry

The alkalinity of the water layer was precisely adjusted to the experimental design (Table 2). The concentrations of calcium and magnesium were markedly lower at high alkalinities (over 0.4 meq l^{-1}) than at low alkalinities. The concentrations of nutrients such as ammonium, nitrate and sulphate were not influenced by alkalinity or sediment treatments. The ortho-phosphate concentrations were strongly affected ($p < 0.01$) in this experiment (Figure 2), at high alkalinities the phosphate concentration was much higher (0.8 µeq l^{-1}) than at low alkalinities (0.2 µeq l^{-1}). Phosphate concentrations were slightly affected by sediment type: they were mostly higher in the water layer above the organic sediment.

Development of macrophytes

After the end of the culture experiment (15 weeks) the biomass of *Littorella uniflora* and *Juncus bulbosus* was determined. It was found that there had been hardly any growth of these species on mineral sediments (Figure 3). Also on organic sediments, the biomass of *Littorella uniflora* hardly increased and was not influenced by alkalinity. Significant differences in growth at different sediments were found for *J. bulbosus*. The biomass of *J. bulbosus* was strongly affected by alkalinity in combination with organic sediments (Figure 3). At low alkalinities (under 0.4 meq l^{-1}) the biomass

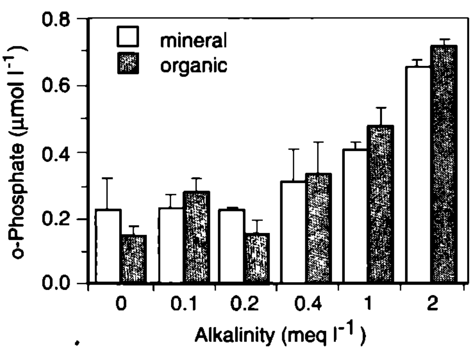


Figure 2: Release of orthophosphate at different alkalinities during the culture experiment.

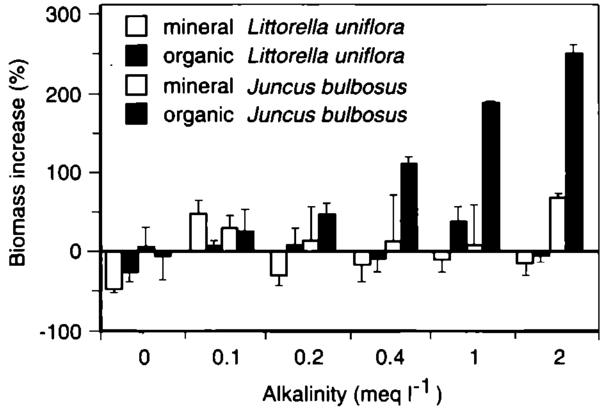


Figure 3: The percent biomass increase of *Juncus bulbosus* and *Littorella uniflora* at different alkalinities during the cultivation experiment (15 weeks).

was less than 50% of the initial biomass, whereas with increasing alkalinity (over 0.4 meq l^{-1}) the biomass increased to more than 250% of the initial biomass ($p < 0.01$).

The seed bank experiment

Different rain treatments

In the first seed bank experiment, independent of the quality of the rain received, several specimens of *Littorella uniflora* germinated in the sediments (Table 3). Some specimens of *Lobelia dortmanna* were found in the clean rain water treatments of the Peetersven sediment. In all sediments, many seedlings of *J. bulbosus*, *J. effusus* L. and *J. bufonius* L. developed. In the sediments of the Beuven more helophytes such as *Phragmites australis* (Cav.) Trin. ex Steudel and *Typha angustifolia* L. developed, in contrast to the Peetersven, where *Agrostis canina* L. became one of the codominating species (Bellemakers *et al.*, 1990).

Liming of sediments

Liming the water layer of the mini-ecosystems led to significant increases of pH and alkalinity. The other parameters showed hardly any changes after treatment of the water layer. The complete comparison of historical data and the results of liming during these experiments were presented by Verheggen (1991).

After liming the mini-ecosystems of the Ven bij Schaijk, Padvindersven and Voorste Goorven, *J. effusus* and *J. bulbosus* became very abundant (Table 4). In mini-ecosystems with organic sediment the abundance of *J. bulbosus* was higher than in those with mineral sediment. In the first year, *Juncus bulbosus* germinated on the shores of the mini-ecosystems; later on this macrophyte colonised the water layer too, as a result of the formation of stolons.

Table 3: The abundance of macrophytes observed in mini-ecosystems with sediments of the Beuven or the Peetersven after two growing seasons. Abundance: 1: one specimen, 2: few specimens, 3: over 10% coverage and 4: over 50% coverage (Bellemakers *et al.*, 1990).

Macrophytes	Beuven		Peetersven	
	Clean	Polluted	Clean	Polluted
Juncus bulbosus L.	2	3	4	4
Sphagnum spec.			2	3
Littorella uniflora (L.) Aschers.	2	2	2	3
Lobelia dortmanna L.			2	

Table 4: Abundance of macrophytes observed in mini-ecosystems with sediments of Ven bij Schaijk, Padvindersven or Voorste Goorven after two growing seasons. For more details see legend Table 3 (Bellemakers *et al.*, 1990).

Macrophytes	Ven bij Schaijk		Padvindersven		Voorste Goorven	
	Mineral	Organic	Mineral	Organic	Mineral	Organic
Juncus bulbosus L.	4	4	4	4	3	3
Sphagnum spec.	2	2	2	2	2	
Littorella uniflora (L.) Aschers.				2		

Littorella uniflora germinated in the Padvindersven sediment and *Sparganium minimum* L. and *Potamogeton polygonifolius* Pourret in the Voorste Goorven sediment. During this experiment no soft water species germinated in the sediment from the Ven bij Schaijk.

Discussion

Effects of liming on the water chemistry of acidified waters

It is necessary to study the effects of liming on water chemistry of acidified moorland pools. In a previous study, Bellemakers and Van Dam (1992) showed that subtle liming of small moorland pools can create slightly buffered conditions which improved the breeding success of the moor frog (*Rana arvalis* Nilsson). Liming experiments on a larger scale showed that each moorland pool has to be thoroughly studied, to ascertain whether liming is an useful method (Bellemakers *et al.*, 1990).

During the culture experiments the water chemistry showed relatively stable characteristics, except for calcium, magnesium and ortho-phosphate (Table 2). An exchange of calcium and magnesium against protons in the sediment may be a possible explanation for the decrease of calcium and magnesium at higher alkalinities in the sediment water interface (Bellemakers *et al.*, 1990).

The significant increase of ortho-phosphate at higher pH and alkalinities, particularly on organic sediments (Table 2), is probably caused by the increased mineralisation and mobilisation of nutrients (Roelofs, 1991). Several studies have shown that following increase in pH and alkalinity, microbial activity in the sediment increases (Brock *et al.*, 1985; Leuven and Wolfs, 1988; Kok and Van de Laar, 1991). This can result in an enhanced phosphate release from the sediment to the water layer (Jackson and Schindler, 1975; Boström *et al.*, 1982; Baccini, 1985).

This proces of internal eutrophication after liming is a serious threat for the restoration of acidified moorland pools. Without creating proper water chemistry conditions (i.e. weakly buffered, oligotrophic, clear water) it is impossible for macrophytes except for *Juncus bulbosus* and some *Sphagnum* species to develop in these waters. Roelofs *et al.* (1994) found after liming of Norwegian lakes a massive development of *Juncus bulbosus* without enhanced nutrient levels in the water layer, except for carbon dioxide. In this experiment, the nitrogen and phosphorus levels in the water layer above the organic sediment had increased very strongly and the macrophyte *Juncus bulbosus* can benefit from these circumstances. Therefore, in shallow lakes it is necessary to remove the organic sapropelium layer, before liming.

Development of submerged macrophytes

Roelofs (1983) and Schuurkes (1987) mentioned the important role of dissolved carbon dioxide and ammonium in the water layer and the sediment pore water in relation to the abundant growth of *J. bulbosus*. In the culture experiment a clear correlation was found between the increase in ortho-phosphate and the explosive growth of *J. bulbosus*, especially on organic sediments. No correlation between nitrogen levels and the explosive growth of *J. bulbosus* was found, probably as a consequence of the high uptake of nitrogen by *J. bulbosus*.

Following liming of Norwegian lakes dense stands of *J. bulbosus* also developed as a result of strongly increased carbon dioxide, ammonium and phosphate levels in the sediment pore water of the organic sediments (Roelofs *et al.*, 1994).

Germination of macrophytes from seed bank material

The sediments of the Beuven and the Peetersven were not limed, but only treated with artificial rain water. In this way, the spontaneous regeneration of macrophytes from those moorland pools under clean conditions could be studied. The germination of *Lobelia dortmanna* and *Littorella uniflora* was remarkable as those soft water species were not observed for a long time in the Peetersven itself. *Lobelia dortmanna* was last observed in the Peetersven in 1962 and *Littorella uniflora* in 1969 (Arts, 1990).

Also, *Eleocharis acicularis* (L.) Roemer et Schultes and *Elatine hexandra* (Lapierre) DC were found (Bellemakers *et al.*, 1990). Thus, the seedbank of these acidified moorland pools still persists after almost three decades of acid conditions. Similar observations were made in the experiments with seed-bank material from the Ven bij Schaijk, the Padvindersven and the Voorste Goorven (Table 4). In the Padvindersven sediment, a specimen of *Littorella uniflora* developed. This species had not been observed here since 1957 (Arts, 1990). This means that the original Littorellion vegetation can be restored, if the right conditions are created. It can be concluded from the germination experiments that there are possibilities for the restoration of these acidified moorland pools.
In the field situation, it was obvious that *J. bulbosus* dominated the water layer after acidification (Roelofs, 1983). In contrast to *J. bulbosus*, the growth rate of *Littorella uniflora* and other isoetid plant species is slow. Earlier experiments by Roelofs *et al.* (1984) clearly showed that *Littorella uniflora* can persist in acidified and nutrient-enriched waters, when *J. bulbosus* is not present. The combination of these two species in acidic waters with high nutrient levels always leads to the dominance of *J. bulbosus*, due to the strong competitive power of the latter.
The abundant growth of *J. bulbosus* was also observed in a whole-lake liming experiment without removal of the sapropelium layer in the Ven bij Schaijk (Bellemakers *et al.*, 1996).

Buffering of acidified moorland pools

Restoration management of moorland pools seems necessary, because control of acidic deposition will still take decades to achieve. To restore the communities of acidified moorland pools, it is necessary to restore the original water chemistry, but it is also necessary to carry out additional restoration measures such as the addition of buffering substances or buffered water and removal of the organic sapropelium layer.
Removal of the organic material will also remove large amounts of nutrients, as well as the latest seedbank. There are still vital seeds in the sediments of acidified lakes, even of species that disappeared many years ago.
As a follow-up of this study, several whole-lake experiments have been conducted, in which the impact of liming and removal of the sapropelium layer of some Dutch moorland pools were investigated (Bellemakers *et al.*, subm.)

Acknowledgements

The authors are indebted to Dr H van Dam, Dr R Bobbink and Prof Dr C den Hartog for critically reading and improving the manuscript We also thank Hanske van Westreenen and Jan Peters from the illustrations department of the Faculty of Sciences for preparing the drawings for this paper and G M Verheggen for technical assistance during chemical analysis We also thank the students M van Gameren and J W P Geenen for practical assistance in the field and experimental work This study was financed by the Ministry of Housing, Physical Planning and Environment, Directorate Water (Project 011661)

References

Arts, G H P, 1990 Deterioration of atlantic soft-water systems and their flora, a historical account Thesis University of Nijmegen, 197 pp

Baalsrud, K , Hindlar, A , Johannessen, M and Matzow, D , 1985 Liming of acid water Report of Department of the Environment (Norway), Directorate for Nature Management, 147 pp

Baccini, P, 1985 Phosphate interactions at the sediment-water interface In W Stumm (Ed), Chemical Processes in Lakes Wiley, New York, p 189-224

Bellemakers, M J S, Maessen, M and Verheggen, G M , 1990 Restauratie van verzuurde en geeutrofieerde zwak gebufferde oppervlaktewateren, mogelijkheden tot herstel Vakgroep Aquatische Oecologie and Biogeologie, University of Nijmegen , i o v D G M /Dir /Water, Ministry van V R O M , 97 pp (in Dutch)

Bellemakers, M J S and Van Dam, H , 1992 Improvement of breeding success of the moor frog (*Rana arvalis*) by liming of acid moorland pools and the consequences of liming for water chemistry and diatoms Environ Pollut 78 165-171

Bellemakers, M J S , Maessen, M and Roelofs, J G M , 1994 Effects of liming on water chemistry in shallow acidified pools in the Netherlands enclosure experiments Water, Air, Soil Pollut 73 131-142

Bellemakers, M J S , Maessen, M , Bobbink, R and Roelofs, J G M , subm Restoration measures against effects of acidification and eutrophication of shallow surface waters in the Netherlands Restor Ecol

Bostrom, B , Jansson, M and Forsberg, C , 1982 Phosporus release from lake sediments Arch Hydrobiol Beih /Ergebn Limnol 18 5-60

Brock, Th C M , Boon, J J and Paffen, B G P, 1985 The effects of the season and of water chemistry on the decomposition of *Nymphaea alba* L , weight loss and pyrolysis mass spectrometry of the particulate matter Aquat Bot 22 197-229

Brown, J M and Goodyear, C D , 1987 Acid Precipitation Mitigation Program research methods and protocols U S Fish Wild Serv National Ecology Center, Leetown, WV NEC-87/27 102 pp

Buskens, R F M , 1989 Beuven herstel van een oecosysteem Report Department Aquatic Ecology and Biogeology, University of Nijmegen, in order of the Ministry of Agriculture, Nature Conservation and Fisheries, 135 pp (in Dutch)

Den Hartog, C., 1993. Effectgerichte maatregelen tegen verzuring en eutrofiëring in natuurterreinen. In: Cals, M.J.R., De Graaf, M.C.C. and Roelofs, J.G.M. (Eds), Effectgerichte maatregelen tegen verzuring en eutrofiëring in natuurterreinen, University of Nijmegen, p. 1-5 (in Dutch).

Drabløs, D. and Tollan, A. (eds), 1980. Ecological impact of acid precipitation. Proc. Int. Conf. Ecol. Impact Acid Precip., Norway 1980, SNSF-project, Oslo-Aas 1980, 283 pp.

Grasshoff, H., and Johannsen, H., 1977. A new sensitive method for the determination of ammonia in seawater. Water Res. 2: 516.

Henriksen, A., 1965. An automated method for determining low-level concentrations of phosphate in fresh and saline waters. Analyst (London) 90: 29-34.

Henriksen, A., 1980. Acidification of freshwaters - a large scale titration. In: Drablös, D. and Tollan A. (Editors), Proceedings of the International Conference on the Ecological Impacts of Acid Precipitation. Oslo, Norway: SNSF project, p. 68-74.

Hultberg, H. and Anderson, I., 1982. Liming of acidified lakes: induced long-term changes. Water, Air, Soil Pollut. 18: 311-331.

Jackson, T.A. and Schindler, D.W., 1975. The biogeochemistry of phosphorus in an experimental lake environment: evidence for the formation of humic-metal-phosphate complexes. Verh. Internat. Verein. Limnol. 19: 211-221.

Kamphake, L.H., Hannah, S.A. and Cohen, J.M., 1967. Automated analysis for nitrate by hydrazine reduction. Water Res. 1: 206.

Kok C.J. and Van de Laar, B., 1991. Influence of pH and buffering capacity on the decomposition of *Nymphaea alba* L. detritus in laboratory experiments: a possible explanation for the inhibition of decomposition at low alkalinity. Verh. Internat. Verein. Limnol. 24: 2689-2692.

Leuven, R.S.E.W., 1988. Impact of acidification on aquatic ecosystems in the Netherlands with emphasis on structural and functional changes. Thesis University of Nijmegen, 181 pp.

Leuven, R.S.E.W. and Wolfs, W.J., 1988. Effects of water acidification on the decomposition of *Juncus bulbosus* L. Aquat. Bot. 31: 57-81.

Muniz, I.P., 1991. Freshwater acidification: its effects on species and communities of freshwater microbes, plants and animals. In: Last, F.T. and Watling, R. (Editors): Proc. Roy. Soc. Edinburgh, 97B: 227-254.

O'Brien, J., 1962. Automatic analysis of chlorides in sewage wastes. Engineering 33: 670-672.

Roelofs, J.G.M., 1983. Impact of acidification and eutrophication on macrophyte communities in soft waters in the Netherlands. I. Field observations. Aquat. Bot. 17: 139-155.

Roelofs, J.G.M., Schuurkes J.A.A.R. and Smits, A.J.M., 1984. Impact of acidification and eutrophication on macrophyte communities in soft waters. II. Experimental studies. Aquat. Bot. 18: 389-411.

Roelofs, J.G.M., 1991. Inlet of alkaline river water into peaty lowlands: effects on water quality and *Stratiotes aloides* L. stands. Aquat. Bot. 39: 267-293.

Roelofs, J.G.M., Brandrud, T.E. and Smolders, A., 1994. Mass invasion of *Juncus bulbosus* after liming of acidified Norwegian lakes. Aquat. Bot. 48: 187-202.

SAS Institute Inc., 1990[a]. SAS Language: Reference, Version 6, First Edition. Sas Institute Inc. Cary NC. USA, 1042 pp.

SAS Institute Inc , 1990[b] SAS/STAT Users Guide, Version 6, Fourth Edition, Volume 1 and 2 Sas Institute Inc Cary NC USA, 1686 pp

Schindler, D W, Frost, T M , Mills, K H , Chang, PS S , Davies, I J , Findlay,, L Malley, D F, Shearer, J A , Turner, M A , Garrison, PJ , Watras, C J , Webster, K , Gunn, J M , Brezonik, PL and Swenson, W A , 1991 Comparisons between experimentally- and atmospherically acidified lakes during stress and recovery In Last, FT and Watling R (Editors) Proceedings of the Royal Society of Edinburgh, 97B 193-226

Schofield, C L , Gloss, S P and Josephson, D , 1986 Extensive evaluation of lake liming, restocking strategies, and fish population response in acidic lakes following neutralization by liming U S Fish and Wildlife Service, Eastern Energy and Land Use Team Interim Progress Rept NEC-86/18, 117 pp

Schoof - Van Pelt, M M , 1973 Littorelletea a study of the vegetation of some amphiphytic communities of Western Europe Thesis University of Nijmegen, 216 pp

Schuurkes, J A A R , 1987 Acidification of surface waters by atmospheric deposition Thesis University of Nijmegen, 160 pp

Sokal, R R and Rohlf, F J , 1981 Biometry, second edition Freeman, San Francisco, 859 pp

Technicon Auto-Analyzer Methodology, 1981 Industrial Method 635-81W, New York

Van Dam, H , Suurmond, E and Ter Braak, C J F, 1981 Impact of acidification on diatoms and chemistry of Dutch moorland pools Hydrobiologia 83 425-459

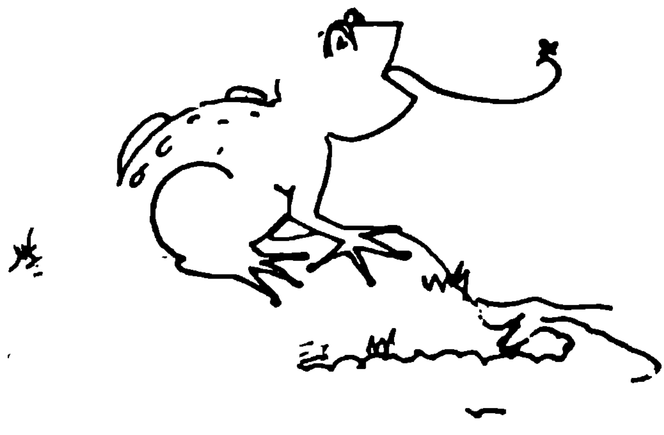
Van Dam, H , Mertens, A and Bellemakers, M J S , 1988 Effects of liming on attached diatoms in experimental enclosures in an acidified moorland pool (preliminary results) Proceedings Third International Conference on the Conservation and Management of Lakes Balaton '88 September 11-17, Keszthely, Hungary, p 51

Verheggen, G M , 1991 Herstel van de oorspronkelijke vegetatie in verzuurde vennen Report no 314 Laboratory of Aquatic Ecology and Biogeology, University of Nijmegen, 32 pp (in Dutch)

Westhoff, V and Den Held, A J , 1973 Plantengemeenschappen in Nederland Thieme and Cie Zutphen, 417 pp (in Dutch)

5

Improvement of Breeding Success of the Moor Frog (*Rana arvalis*) by Liming of Acid Moorland Pools and the Consequences of Liming for Water Chemistry and Diatoms



Abstract

Liming experiments with powdered limestone (grain size < 3 mm) were conducted in eight acid shallow moorland pools in the Tongerense Heide heathland area from February 1988 until November 1989 The effects on water chemistry, diatoms and fungal infection of the eggs of the moor frog were studied After an initial treatment in March 1988 the pH increased from ca 4 0 to ca 5 0 in those pools which dried out in summer and to ca 6 0 in the permanent pools Alkalinity increased from 0 to 20 - 200 meq l ^{1} in temporary pools and from 0 to 300 - 500 meq l ^{1} in permanent pools As drying out of the pools caused reacidification, after refilling the temporary pools were relimed in March 1989 No significant changes were found in concentrations of phosphate and nitrogen compounds *Eunotia paludosa*, which is characteristic for oligotrophic, very acid pools and bogs with a fluctuating water table, was the dominant diatom in the untreated pools It was replaced by eutraphentic and saprophilous taxa, particularly in the permanent pools Species from extremely soft waters, which are very sensitive to acidification, were found only occasionally in some samples from the treated permanent pools After liming the proportion of infected moor frog eggs decreased from ca 95% in the untreated to ca 5% in the treated pools

M J S Bellemakers and H van Dam, 1992 Improvement of breeding success of the moor frog (*Rana arvalis*) by liming of acid moorland pools and the consequences of liming for water chemistry and diatoms Environmental Pollution 78 165-171

Introduction

The adverse impact of acidification by atmospheric deposition on biota of soft shallow waters in The Netherlands is well documented (*e.g.* van Dam *et al.*, 1981; Roelofs, 1983; Leuven *et al.*, 1986[b]). These studies demonstrate a severe decline in the diversity of a number of major groups of organisms, including amphibians.

Leuven *et al.* (1986[a]) demonstrated that the eggs of the moor frog (*Rana arvalis* Nilsson) were heavily infected with fungi at spawning sites in very acid waters in The Netherlands. In waters with a pH of 4.2, 50% of the eggs died. Clausnitzer (1979) showed the dramatic decrease of the moor frog as a result of acidification in adjacent areas of Germany.

As the moor frog is a threatened species in Europe, with its optimum occurrence in naturally acid moorland and heathlands of The Netherlands, Belgium and Germany, conservation measures are needed (Strijbosch, 1979; Corbett, 1989). Therefore, liming experiments were conducted to examine the possibility of improving the breeding success of the moor frog in an area with a number of shallow moorland pools the main habitat of this species in The Netherlands.

The aim of this study was to find a method of improving the breeding success of the moor frogs with minimal damage to other biota. As the water of moorland pools is poor in lime, adverse changes in chemistry may be expected with the addition of lime: increased pH-levels may invoke the release of nutrients by enhanced decomposition of sedimented organic material (Carpenter, 1980; Roelofs, 1991). This internal eutrophication will affect the biota. Therefore, the changes of the chemical composition of most important ions and nutrients, the composition of the vegetation of macrophytes (the most important primary producers), and the species composition of epiphytic assemblages of diatoms were investigated. These algae are excellent indicators of water acidity and trophic state (*e.g.* Kalbe, 1973; Charles *et al.*, 1989).

Materials and Methods

Study sites and design of experiments

The moorland pools which were studied are situated on the heathlands of the nature reserve Tongerense Heide, in the central part of The Netherlands (52° 20 N, 5° 55 E). As the pools have perched water tables and their drainage area is hardly larger than their surface area, they are exclusively fed by rain water.

The vegetation of the heathland area is dominated by *Calluna vulgaris* (L.) Hull, *Empetrum nigrum* L. and *Erica tetralix* L. Comparison with inventories made in 1952 and in 1962 reveals that the quantity of *Molinia caerulea* (L.) Moench has increased in the last few decades, probably due to acid atmospheric deposition (Heil and Diemont, 1983). The vegetation of the pools is dominated

Figure 1: Liming of the pools with marlstone. Behind the pool there are some heaps of removed *Sphagnum*.

by *Eriophorum angustifolium* Honckeny, *Carex rostrata* Stokes and submerged bog-mosses (*Sphagnum* spec.) and is virtually unchanged since the inventory by Van Zeist (1946). More details of the study sites, methods and results are presented by Luttikholt (1989). The fungal infection rate of the moor frog eggs in the pools of the area has increased over the last decade (A.J.M. Roozen, pers. comm.).

Eight pools, where egg deposition of moor frogs had been observed in previous years, were selected for study (Table 1). In two pools, powdered limestone (Emkal, grain size < 3 mm) was added. In another two pools *Sphagnum* and most of the organic sediment was removed prior to the addition of limestone, in order to prevent eutrophication of the pools by mineralization of the organic material after liming (Figure 1). Four pools were not limed: two were used as a control and in two pools *Sphagnum* was removed. Each of the treatments included an intermittent pool and a permanent

Table 1: Treatments and morphometric characteristics of investigated pools.

Pool	Area $10^3 m^2$	Mean depth (cm)	Limestone added in March'88 (kg)	Limestone added in March'89 (kg)
Ci	1.0	25	0	0
Cp	3.1	43	0	0
Si	0.2	13	0	0
Sp	1.5	20	0	0
Li	0.8	16	22.5	22.5
Lp	0.6	40	24.0	0
SLi	0.5	19	15.0	15.0
SLp	1.5	32	48.0	0

C=control; S=*Sphagnum* removed; L=limestone added; i=intermittent and p=permanent pool, which dried out in extremely dry summers only. The permanent pools were limed once, the intermittent pools were relimed.

Field and laboratory procedures

Water samples for chemical analysis were taken from the centre of the pools at least every 2 months, between March 1988 and November 1989. pH and alkalinity were measured within two hours after sampling. pH was measured with a GK2501B combined pH electrode, connected to a Radiometer Copenhagen PHM82 pH/mV meter. Alkalinity was determined by titration of a subsample with 0.01 N HCl down to pH 4.2. Samples for analysis in the laboratory were passed through a Whatman GF/C filter (1.2 μm), fixed by addition of mercury (II) chloride, stored in iodated polyethylene bottles and frozen at -28 °C until chemical analysis.

Chemical analysis of water samples were carried out according to Technicon Auto-analyzer Methodology (1981). Calcium was analysed by the Inductive Coupled Plasma Method using an Instrumentation Laboratory Plasma 2000. Water chemistry data were statistically compared using the signed-rank Wilcoxon test (Sokal and Rohlf, 1981).

Water depth was recorded every 2 months in the centre of each pool between March 1988 and November 1989. Diatoms were sampled simultaneously at the same spot, by squeezing *Sphagnum* and Cyperaceae. Old samples were searched for in university collections. Slides were prepared for microscopic examination and the percentage abundance of each taxon was assessed after counting 200 valves in each slide. Details of the methods and keys used for identification of diatoms are given by Van Dam and Mertens (1990). Autecological data for the diatom taxa were borrowed from numerous publications.

In the spawning period (March - May) of the years 1987, 1988 and 1989 the percentage fungal infection of the eggs of the moor frog was estimated by visual inspection of the entire pools every two weeks.

Results

Water chemistry

The effects of liming on water chemistry are shown in Table 2 and Figure 2. After liming, pH and alkalinity increased significantly ($p < 0.01$). There was a significant increase in the ammonium and calcium concentrations in the water of the limed pools, especially for ammonium when *Sphagnum* was removed (Table 2). In the permanent pools the pH rose to 7 and the alkalinity attained maximum values of 400-500 $\mu eq\ l^{-1}$ just after liming; in limed intermittent pools, the increase was much less (pH 5-6.5, alkalinity 200 $\mu eq\ l^{-1}$). After drying out and refilling with rain water, the pH and alkalinity of the limed temporary pools (Li and SLi) returned to pretreatment levels.

Table 2: Geometric mean values of selected chemical variables.

Pool	pH	Alkalinity µeq l⁻¹	PO_4^{3-} µmol l⁻¹	NH_4^+ µmol l⁻¹	NO_3^- µmol l⁻¹	SO_4^{2-} µmol l⁻¹	Ca^{2+} µmol l⁻¹
Ci	4.1	0	0.40	6.0	2.0	98	21
Cp	4.1	0	0.34	5.5	3.4	84	28
Si	3.9	0	0.31	6.2	2.9	100	28
Sp	4.0	0	0.43	9.0	3.9	120	22
Li	4.6	26	0.28	5.2	2.7	88	46
Lp	5.6	95	0.21	3.9	3.3	96	43
SLi	4.6	42	0.37	10.4	3.2	121	42
SLp	5.4	112	0.46	11.4	2.4	133	45

C=control; S=*Sphagnum* removed; L=limestone added; i=intermittent and p=permanent

Concentration of the nutrients phosphate and nitrate were not affected by liming and remained at low levels throughout the experiments. The removal of *Sphagnum* did not affect the surface water chemistry, irrespective of liming.

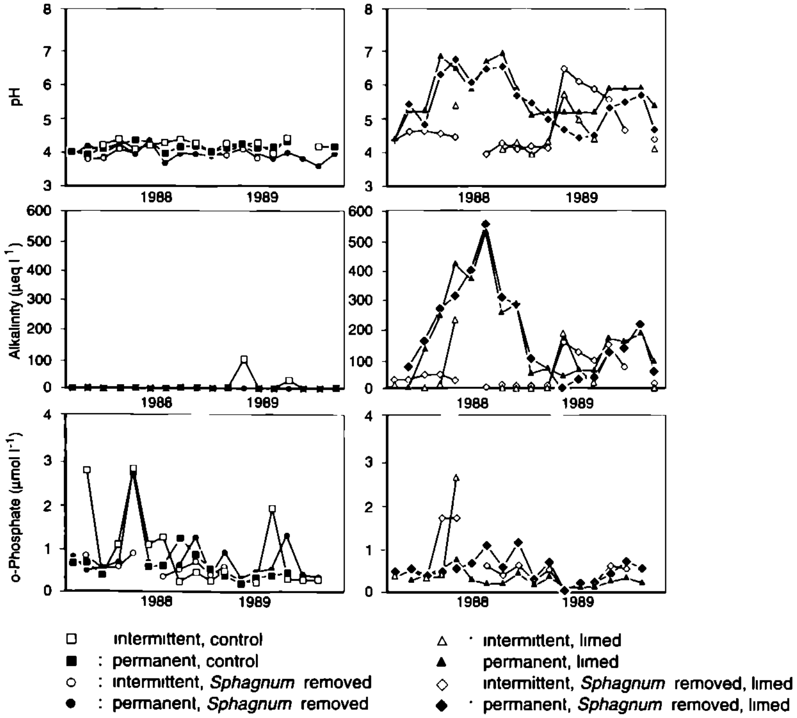


Figure 2: Changes in pH, alkalinity and phosphate concentrations during the experiments from all pools.

Diatoms

In total 32 diatom taxa were observed. The 16 taxa with a mean percent abundance > 2% in at least one pool are listed in Table 3, which includes also the results of two samples from some pools in the Tongerense Heide, taken by O.H. Westerhof in the summer of 1932. As all pools in this area are very similar, these samples provide some information on the earlier condition of the recently investigated pools.

We have distinguished five ecological groups of diatoms. The acid-water taxa are wide-spread taxa from extremely acid sites; soft-water taxa are rarer taxa from weakly buffered sites and threatened by acidification; eurytopic taxa are found in a wide variety of water types; eutraphentic taxa are typical for eutrophic, often alkaline, water bodies, and saprophilous taxa are found in moderately acid to alkaline water, loaded with biodegradable organic material.

The acid-water taxa comprise over 90% (in many cases even 100%) of all the taxa in the samples from control pools and the pools where only *Sphagnum* was removed (Table 3; Figure 3). Only in the permanent pool where the bog-mosses were removed were some soft-water taxa present. The

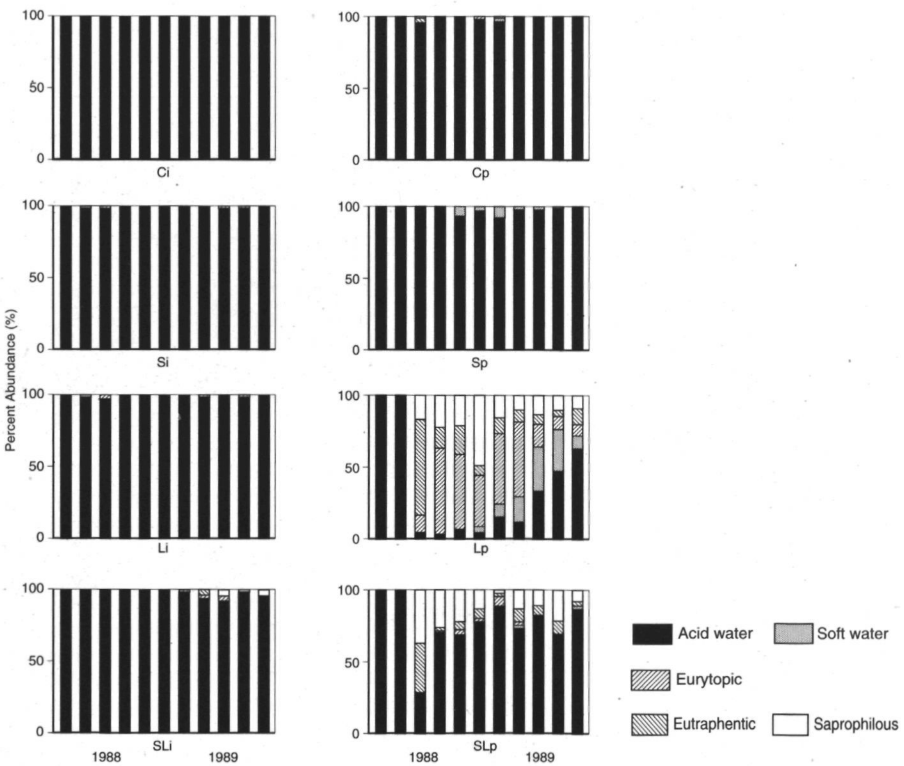


Figure 3: Percent abundage of ecological groups of diatoms in samples from 1988 and 1989 from all pools. C=control; S=*Sphagnum* removed; L=limestone added; i=intermittent and p=permanent.

Table 3: Mean percent abundance of most important diatom taxa in 1932 (n=2) and from May 1988 - November 1988 (n=10). Taxa are classified according tot ecological groups.

pH-opt.	pH-group	Taxon	Pool: 1932	Ci	Cp	Si	Sp	Li	Lp	SLi	SLp
4.8	cir	*Eunotia bilunaris* (Ehr.) Mills .	16	66	15	0	20	27	14	39	46
3.8	acb	*E. paludosa* Grun.	34	19	9	80	71	60	2	16	14
4.2	acb	*Frustulia rhomboides* var. *saxonica* Rab.	38	9	51	13	4	6	11	27	7
4.2	acb	*F. rhomboides* var. *crassinerva* (Bréb.) Ross	-	5	21	4	2	4	1	11	1
—	acp	*Pinnularia divergentissima* (Grun.) Cl.	4	-	3	0	-	-	0	-	4
—	acb	*P. subcapitata* var. *hiseana* (Jan.) Mull.	3	1	0	2	1	3	-	3	1
—	—	others	1	0	0	1	0	1	0	0	0
Subtotals acid-water taxa			96	100	99	100	98	100	28	96	74
4.2	acp	*Eunotia naegelii* Migula	-	-	0	-	3	-	1	0	-
—	acp	*E. pectinalis* var. *minor* (Kütz.) Rab.	1	-	0	-	-	-	2	-	0
5.0	acp	*Tabellaria flocculosa* (Roth) Kütz.	-	-	0	-	-	-	7	0	0
—	—	others	-	-	-	0	-	0	0	-	-
Subtotals soft-water taxa			1	0	1	0	3	0	10	0	0
6.8	cir	*Achnanthes minutissima* Kütz.	3	-	0	-	-	1	29	1	1
—	—	others	1	0	0	0	0	-	0	0	1
Subtotals eurytopic taxa			4	0	0	0	0	1	29	1	1
—	alp	*Fragilaria capucina* Desmaz.	-	-	0	-	-	-	15	1	7
—	—	others	-	-	-	-	-	0	-	0	1
Subtotals eutraphentic taxa			-	-	0	-	-	0	15	1	8
—	cir	*Gomphonema gracile* Ehr.	-	-	0	-	-	-	10	1	1
5.1	cir	*G. parvulum* Kütz	-	-	0	-	-	-	6	1	10
—	cir	*Nitzschia archibaldii* Lange-Bert.	-	-	-	-	-	-	0	0	2
—	cir	*N. palea* (Kütz.) Smith	-	-	-	-	-	0	0	0	2
—	cir	*N. palea* var. *debilis* (Kütz.) Grun.	-	-	-	-	-	-	-	-	2
Subtotals saprophilous taxa			-	-	0	-	-	0	16	2	17

pH-opt. = pH-optimum according to Ter Braak and Van Dam (1989). pH-group=classification in pH-system according to Hustedt (1939). (acb=acidobiontic, acp=acidophilous, cir=circumneutral, alp=alkaliphilous). -=not found, 0=percentage abundance < 0.5%. C=control; S=*Sphagnum* removed; L=limestone added; i=intermittent and p=permanent.

diatom assemblages from these four pools were very similar to those from the pools which were sampled in 1932. Also in the limed intermittent pools the acid-water taxa made up over 90% of the total. Only in the samples from summer 1989 do eutraphentic and saprophilous taxa occur more than occasionally. In contrast, in the diatom assemblages from the limed permanent pools the dominance of the acid-water taxa changed to a dominance of eurytopic, eutraphentic and saprophilous taxa, particularly in the pool where the bog-mosses were not removed. In the course of the year, the acid-water taxa did increase again, more or less concurrently with the decrease of pH (Figure 2).

Moor frogs

The fungal infection rate of the eggs of the moor frog was 75-100% in the pretreatment year 1987 and in the unlimed pools. The infection rate decreased to 0-25% in the limed pools. Removal of *Sphagnum* did not affect the infection rate of the eggs at all. Moreover, no differences were found between temporary and permanent pools (Figure 4).

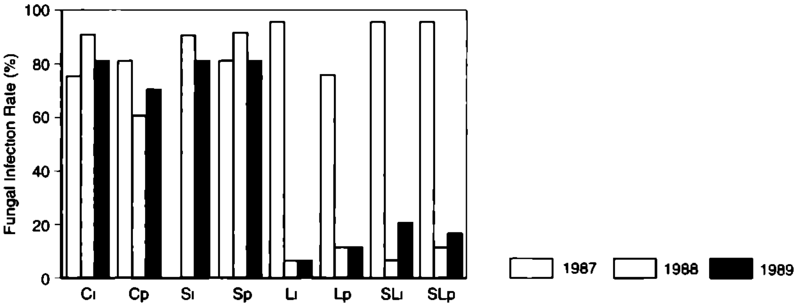


Figure 4: Fungal Infection Rate of the eggs of the moor frogs in different pools and years. C=control; S=*Sphagnum* removed; L=limestone added; i=intermittent and p=permanent.

Discussion

The increase of pH, alkalinity and calcium concentration was also observed in other limed moorland pools (Bellemakers *et al.*, 1994). By using coarse, slowly weathering limestone (marlstone) it was possible to avoid shock effects of pH-increase, such as high concentrations of nutrients due to internal eutrophication by decomposition of detritus and bog-mosses. A gradual dying-off of bog-mosses was observed in the permanent, limed pool. No other conspicuous changes of the vegetation were observed.

As ammonium has accumulated in submerged bog-mosses by atmospheric deposition, release of ammonium after liming was expected (Scheffer and Schachtschabel, 1979; Schuurkes, 1986). Also, in a Swedish limed lake, release of ammonium was observed along with decomposition of *Sphagnum*, but concentrations of ammonium remained low, probably due to denitrification (Broberg, 1987, Dickson, 1989) Therefore, the insignificant increase of ammonium in the studied pools does not necessarily indicate slow mineralization of accumulated organic material Unchanged concentrations of phosphates after liming, as in our pools, seem to be a general feature, although some studies indicate an increase of phosphate concentrations after liming (Bostrom *et al*, 1982, Broberg, 1987, Molot *et al*, 1990, Bellemakers *et al*, 1994)

The permanent pools were limed once The pH and alkalinity rose to relatively high levels in 1988 Probably, the internal generation of alkalinity in a reducing environment was promoted after the initial lime treatment (Schindler, 1986, Curtis, 1989) Afterwards, pH and alkalinity decreased slightly, due to acidification by atmospheric deposition (Van Dam *et al*, 1981, Schuurkes *et al*, 1988)

After drying out and refilling of the temporary moorland pools, pH, alkalinity and calcium concentration decreased significantly, due to dry atmospheric acid deposition and the oxidation of sulfide ions in the aerated dry soil of these pools (Vangenechten *et al*, 1981, Van Dam, 1988)

The diatom assemblages were very poor in species and similar to those of temporary acid waters and raised bogs in Europe The principal species, *Eunotia paludosa* Grun , is a widespread typical aerophilous *Sphagnum* bog species, which occurs mainly in very acid water (pH 3 5-4 5) The other dominant species in all but the limed permanent pools are also commonly found in acid bogs and associated small water bodies (*e g* Petersen, 1950, Schluter, 1961, De Vries, 1984) *Eunotia exigua* (Bréb) Rab , which is often the dominant species in acidified moorland pools and lakes (Van Dam *et al*, 1981, Arzet *et al*, 1986, Beebee *et al*, 1990) was recorded only in low numbers in these samples and is not able to survive the frequent drying out of the pools

The increase in eutraphentic and saprophilous taxa, as in the limed permanent pools (Table 3, Figure 3), has not been observed after liming of acidified lakes (Simola, 1986, Ohl *et al*, 1990, Round, 1990), probably due to a dilution effect Their increase indicates a release of nutrients by the decay of sedimentation organic material and the improved availability of non-refractory organic material for growth of saprophilous diatoms Moreover, the gradual dying-off of the bog mosses in the limed, permanent pool was a supplementary source of nutrients The floristic composition of diatom assemblages indicates a clear increase in the bioavailability of nutrients due to mineralization of accumulated organic material, although the chemical analysis did not show increased nutrient concentrations

The low abundance of eutraphentic and saprophilous taxa in the limed intermittent pools may be due to the much lower peak of pH after liming than in the permanent pools and its return to pre-treatment levels after drying up in summer In the spring of 1989, when the pH of limed, permanent pools, where *Sphagnum* was removed had a higher maximum than in 1988 (Figure 2) the relative abundance of saprophilous and eurytopic diatoms was also higher than in 1988

The increase in eutraphentic and saprophilous taxa is a departure from the base-line situation, as such taxa were not found in the samples from 1932 (Table 3) From the conservationists point of view, the apparent increase of nutrient availability in originally oligotrophic ecosystems is undesirable (Ratcliffe, 1977, Rijksinstituut voor Natuurbeheer, 1979)

The average diatom-inferred pH, calculated as weighted average of the pH-optimum values of the taxa (Ter Braak and Van Dam, 1989) is 4 2 (± 0 7) for the historical samples and 4 2-4 5 (± 0 7) for the samples from the control pools, which would suggest no or only a slight acidification of these pools The similarity of the present macrophyte vegetation of the pools with that recorded by Van Zeist (1946) also does not suggest severe acidification However, the increase of the rate of fungal infection of the moor frog eggs over the last decade does indicate acidification of the pools It has been shown by Leuven *et al* (1986[a]) that the rate of fungal infection increases very rapidly from 25% to 100% when the pH decreases from 4 5 to 3 5 Moor frog populations in this study apparently did not develop adaptations to acid conditions, as was observed in acidified Swedish pools by Andrén *et al* (1989)

The results of this investigation indicate that it is possible to improve the breeding success of the moor frog by liming acid pools Particularly in permanent pools, liming causes undesirable eutrophication by mineralization of organic material The eutrophication is relatively low in intermittent pools, even if the stock of organic material, mainly bog-moss, has not been removed Liming should be repeated every year before oviposition of the moor frog As the long-term effects of such actions are still unknown, monitoring of moor frog populations, water chemistry and biological indicators of trophic state (*e g* macrophytes and diatoms) is necessary

Acknowledgements

We are indebted to Mrs G M Verheggen, Mrs A J G Luttikholt, Mrs A Mertens and Mr J A Sinkeldam for technical assistance The Gueldrian Landscape Foundation permitted us to use their properties for the experiments and its workers removed the bog-mosses from some of the pools Drs A J M Roozen and Mr P Schermerhorn made available their observations on the fungal infection rate of the moor frog Dr F Bouman and Mrs H Kooijman (Department of Biology, University of Amsterdam) put historical diatom samples at our disposal Prof Dr C den Hartog, Dr J G M Roelofs, Dr A W Boxman and Drs M Maessen commented on the manuscript Mrs M W A Melis prepared the drawings for this paper A part of this research was funded by the Ministry of Housing, Physical Planning and Environment, Directorate Water (Project 011661)

References

Andrén, C , Mården, M and Nilson, G , 1989 Tolerance to low pH in a population of moor frogs, *Rana arvalis*, from an acid and a neutral environment a possible case of rapid evolutionary response to acidification Oikos 56 215-223

Arzet, K , Krause-Dellin, D and Steinberg, C , 1986 Acidification of four lakes in the Federal Republic of Germany as reflected by diatom assemblages, cladoceran remains and sediment chemistry In Diatoms and lake acidity, J P Smol, R W Battarbee, R B Davis, J Merilainen (eds), Junk, Dordrecht, p 227-250

Beebee, T J C , Flower, R J , Stevenson, A C , Patrick, S T , Appleby, P G , Fletcher, C , Marsh, C , Natkanski, J , Rippey, B and Battarbee, R W , 1990 Decline of the Natterjack Toad *Bufo calamita* in Britain palaeoecological, documentary and experimental evidence for breeding site acidification Biol Conserv 53 1-20

Bellemakers, M J S , Maessen, M and Roelofs, J G M , 1994 Effects of liming on water chemistry in shallow acidified pools in the Netherlands enclosure experiments Water, Air, Soil Pollut 73 131-142

Bostrom, B , Jansson M and Forsberg, C , 1982 Phosphorus release from lake sediments Arch Hydrobiol Beih Ergebn Limnol 18 5-59

Broberg, O , 1987 Nutrient responses to the liming of lake Gårdsjon Hydrobiologia 150 11-24

Carpenter, S R , 1980 Enrichment of lake Wingra, Winsconsin, by submerged macrophyte decay Ecology 61 1145-1155

Charles, D F , Battarbee, R W , Renberg, I , Van Dam, H and Smol, J P , 1989 Paleological analysis of lake acidification trends in North America and Europe using diatoms and chrysophytes In S A Norton, S E Lindberg and A L Page (eds), Soils, aquatic processes, and lake acidification Springer, New York p 207-276

Clausnitzer, H J , 1979 Durch Umwelteinflusse gestorte Entwicklung beim Laich des Moorfrosches (*Rana arvalis* Nilsson) Beitr Naturk Niedersachs 32 68-78

Corbett, K (ed), 1989 The conservation of European reptiles and amphibians Helm, London, 274 pp

Curtis, P J , 1989 Effects of hydrogen ion and sulphate on the phosphorus cycle of a Precambrian Shield lake Nature 337 156-157

De Vries, B J , 1984 Diatom assemblies in some moorland pools in the Drenthian district (The Netherlands) Hydrobiol Bull 18 3-10

Dickson, W , 1989 Liming of lake Gårdsjon Acidification Research in Sweden 8 4-8

Heil, G W and Diemont, W H , 1983 Raised nutrient levels change heathland into grassland Vegetatio 53 113-120

Hustedt, F , 1939 Systematische und okologische Untersuchungen uber die Diatomeeenflora von Java, Bali und Sumatra Arch fur Hydrobiol /Suppl 16 274-394

Kalbe, L , 1973 Kieselalgen in Binnengewassern Ziemsen, Wittenberg Lutherstadt 206 pp

Leuven R S E W , den Hartog C , Christiaans, M M C and Heijligers, W H C , 1986[a] Effects of water acidification on the distribution pattern and the reproductive success of amphibians Experientia 42 495-503

Leuven, R.S.E.W., Kersten, H.L.M., Schuurkes, J.A.A.R., Roelofs, J.G.M. and Arts, G.H.P., 1986[b]. Evidence for recent acidification of lentic soft waters in The Netherlands. Water, Air, Soil Pollut. 30: 387-392.

Luttikholt, A.J.G., 1989. Effecten van bekalking op diatomeeën in verzuurde vennen op de Tongerense Heide (gemeente Epe). Rijksinstituut voor Natuurbeheer. Wageningen, Leersum. Intern Rapport, 89/29, 50 pp. (in Dutch).

Molot, L.A., Dillon, P.J. and Booth, G.M., 1990. Whole-lake and nearshore water chemistry in Bowland lake, before and after treatment with $CaCO_3$. Can. J. Fish. Aquat. Sci. 47: 412-421.

Ohl, L.E., Gont, R.A. and Dibble, E.D., 1990. Diatom response to liming of a temperate, brown water lake. Can. J. Bot. 68: 347-353.

Petersen, J.B., 1950. Observations on small species of *Eunotia*. Dansk Bot. Ark. 14: 1-19.

Ratcliffe, D., (Ed.), 1977. A nature conservation review. I. Cambridge University Press, Cambridge, 338 pp.

Rijksinstituut voor Natuurbeheer, 1979. Natuurbeheer in Nederland: Levensgemeenschappen. Pudoc, Wageningen, 392 pp. (in Dutch).

Roelofs, J.G.M., 1983. Impact of acidification and eutrophication on macrophyte communities in soft waters in the Netherlands. I. Field observations. Aquat. Bot. 17: 139-155.

Roelofs, J.G.M., 1991. Inlet of alkaline river water into peaty lowlands: Effects on water quality and *Stratiotes aloides* L. stands. Aquat. Bot. 39: 267-293.

Round, F.E., 1990. The effect of liming on the benthic diatom populations in three upland Welsh lakes. Diatom Research 5: 129-140.

Scheffer, F. and Schachtschabel, P., 1979. Lehrbuch der Bodenkunde. Enke, Stuttgart, 394 pp.

Schindler, D.W., 1986. The significance of in-lake production of alkalinity. Water, Air, Soil Pollut. 30: 931-944.

Schuurkes, J.A.A.R., 1986. Atmospheric ammonium sulphate deposition and its role in the acidification and nitrogen enrichment of poorly buffered aquatic systems. Experientia 42: 351-357.

Schuurkes. J.A.A.R., Jansen, J. and Maessen, M., 1988. Water acidification by addition of ammonium sulphate in sediment water columns and in natural waters. Arch. Hydrobiol. 112: 495-516.

Schlüter, M., 1961. Die Diatomeeen-Gesellschaften des Naturschutzgebietes Strausberg bei Berlin. Internat. Rev. ges. Hydrobiol. 462: 231-253.

Simola, H., 1986. Diatom responses to acidification and lime treatment in a clear water lake. comparison of two methods of a diatom stratigraphy. In Diatoms and lake acidity, ed. J. P. Smol, R.W. Battarbee, R.B. Davis, J. Meriläinen. Junk, Dordrecht, p. 221-226.

Sokal, R.R. and Rohlf, F.J., 1981. Biometry, second edition. Freeman, San Francisco, 859 pp.

Strijbosch, H., 1979. Habitat selection of amphibians during their aquatic phase. Oikos 33: 424-431.

Technikon Auto-analyzer Methodology, 1981. Industrial Method 635-81W, New York.

Ter Braak, C.J.F. and Van Dam, H., 1989. Inferring pH from diatoms: a comparison of old and new calibration methods. Hydrobiologia 178: 209-223.

Van Dam, H., Suurmond, G. and Ter Braak, C.J.F., 1981. Impact of acid precipitation on diatoms and chemistry of Dutch moorland pools. Hydrobiologia 83: 425-459.

Van Dam, H , 1988 Acidification of three moorland pools in The Netherlands by acid precipitation and extreme drougth periods over seven decades Freshwat Biol 20 157-176

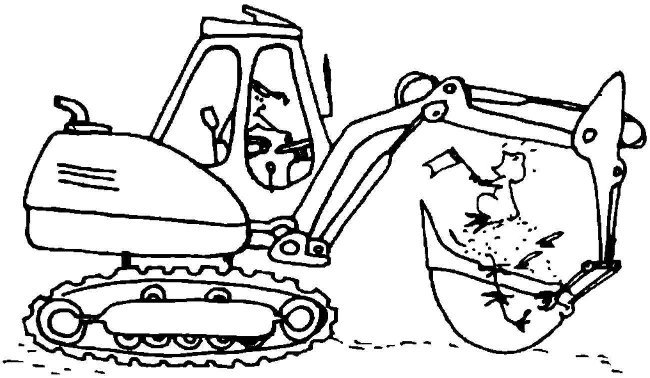
Van Dam, H , Mertens, A , 1990 A comparison of recent epilithic diatom assemblages from the industrially acidified and copper polluted Lake Orta (Northern Italy) with old literature data Diatom Res 5 1-13

Vangenechten, J H D , Bosmans, F , Deckers, H , 1981 Effects of short-term changes in rain water supply on the ionic composition of acid moorland pools in the Campine of Antwerp (Belgium) Hydrobiologia 76 149-159

Van Zeist, W , 1946 Botanische schatkamers op de Veluwe Daar bot Iets 4/5 6-11 (in Dutch)

6

Restoration Measures against Effects of Acidification and Eutrophication in Shallow Surface Waters in the Netherlands



Abstract

In recent decades many soft water macrophytes of the phytosociological class Littorelletea have strongly declined in the Netherlands as a result of acidification and eutrophication processes. Depending on the various types of the water bodies, set out in previous studies, various control measures against the effects of acidification and eutrophication are necessary to restore the original vegetation of the affected surface waters. This study shows that after liming or inflow of buffered water the pH and alkalinity of the pools increased significantly. Without removal of the sapropelium layer, however, re-acidification occurred within a short period. Eutrophication of the system could take place after adding buffering substances due to the increase of pH and alkalinity. Therefore, removal of the nutrient-rich sapropelium layer is necessary to create stable water chemistry conditions, such as pH, alkalinity and the levels of nutrient concentrations. Inflow of buffered water (pre-treated water or ground water) is favourable, since the quantity of inlet water can be easily regulated and there is no accumulation of limestone on the sediments, influencing the biogeochemical cycles in the sediment. The short-term results showed a positive development of water chemistry and vegetation. However, long-term biomonitoring showed that direct liming of weakly buffered moorland pools against acidification caused massive *Juncus bulbosus* L. expansion after the first 2-3 years. Many characteristic soft water macrophytes developed successfully after removal of the sapropelium layer and controlling pH and alkalinity by creating inflow of buffered water. Removal of the sapropelium layer, followed by liming of the water layer does not automatically lead to successful restoration, especially on the long-term. Better results should be achieved by regulated liming techniques, such as controlled inlet of pre-treated water or ground water. Thus, more knowledge of the long-term success of restoration in acidified and eutrophicated shallow surface waters is needed.

M.J.S. Bellemakers, M. Maessen, R. Bobbink and J.G.M. Roelofs. Restoration measures against effects of acidification and eutrophication in shallow waters in the Netherlands. Submitted: Restoration Ecology.

Introduction

Shallow surface waters, mainly fed by precipitation, are vulnerable to processes such as acidification and eutrophication (Drabløs and Tollan, 1980; Overrein *et al.*, 1981). Most of the shallow, oligotrophic, poorly buffered moorland pools in the Netherlands have been acidified by atmospheric deposition of sulphur and nitrogen compounds (Van Dam, 1988; Leuven *et al.*, 1989; Arts, 1990). The pH of many of these pools has decreased and the concentrations of aluminum and sulfate have increased significantly (Schuurkes, 1987; Leuven, 1988). Due to inhibited decomposition at low pH values after acidification of the water layer and the sediment pore water (Rao and Dutka, 1983), a thick, organic sapropelium, consisting of partly decomposed plant material, has developed in most of these pools. Furthermore, intensification of agricultural activities has caused an increase of ammonia/ammonium deposition (Asman, 1987). Nitrification processes also cause a progressing acidification, due to the proton release during the transformation of ammonium into nitrate (Schuurkes *et al.*, 1988). As a result of acidification the characteristic biological communities of these moorland pools have deteriorated (Van Dam *et al.*, 1981; Roelofs, 1983).

In eutrophicated surface waters the increase of nutrients has also led to accumulation of organic matter. The eutrophication of surface waters may be caused by an external supply of nutrients (external eutrophication by waste water, fertilizers, atmospheric deposition and bird faeces), or by release of nutrients via chemical changes in the sediment (internal eutrophication; Mortimer, 1971). As a result of both external and internal eutrophication, the water quality in many surface waters decreased rapidly (Roelofs, 1991).

In the Netherlands endangered soft water macrophytes occur in small, shallow waters in the sandy areas in the eastern, central and southern parts of the country and in the coastal dunes. The waters are weakly buffered and oligotrophic, with fluctuating water levels. *Littorella uniflora* (L.) Aschers. and *Lobelia dortmanna* L. are characteristic representatives of the Littorellion (Schoof-Van Pelt, 1973) and highly adapted to these conditions.

These soft water macrophytes are mainly water plants with an isoetid growth form (Den Hartog and Van der Velde, 1988) and are adapted to extreme circumstances, due to low levels of carbon dioxide, nitrogen and phosphorus in the water layer (Roelofs *et al.*, 1984). The thick stiff leaves of these macrophytes are compact, and cause a reduction of the surface-volume ratio. Further all isoetids have a well-developed system of internal air lacunae, so that carbon dioxide produced during the respiration can be reused again (Søndergaard, 1979). They are able to take up carbon dioxide with the roots from the sediment pore water, where the carbon dioxide level may be 10-100 times higher than in the overlying water layer (Wium-Andersen, 1971; Søndergaard and Sand-Jensen, 1979), and they have a high oxygen release by the roots (Sand-Jensen *et al.*, 1982). This can contribute to the oxidation of organic matter and nitrification of ammonium in the sediment, which provides them with carbon dioxide and nitrate

Furthermore, most of the isoetids apply a special mechanism for photosynthesis, similar to the Crassulacean Acid Metabolism (CAM) This may be interpreted as an adaptation to environments where carbon dioxide availability is precarious (Keeley, 1983) Also, there is a clear correlation between the development of the underground biomass and the nutrient content of the environment, in oligotrophic conditions the root system is well-developed (Sand-Jensen and Søndergaard, 1979) So the isoetids fit functionally very well within the environment where they occur, and this growth form assures the most efficient use of the scarce but essential resources In environments richer in nutrients these plants would not stand a chance in the competition with other more demanding species, which usually can grow more efficient, but faster and dominate over the characteristic soft water species (Roelofs *et al* , 1984, Madsen, 1985, Den Hartog, 1986, Smits *et al* , 1990)

Since the beginning of this century a dramatic decline in the occurrence of these species has taken place as a result of acidification and eutrophication (Arts, 1990) Many sites have been overgrown by *Juncus bulbosus* L or epiphytic algae (Roelofs, 1983) or by succession to other vegetation types (Arts *et al* , 1990)

To restore shallow acidified or eutrophicated surface waters on the long term, a drastic decrease in acidifying and eutrophicating atmospheric deposition is necessary (Bobbink *et al* , 1992) Emission control measures against acidification and eutrophication are not expected to be effective before the next decade, in spite of the encouraging decrease of sulphur dioxide emissions and the recent stability of ammonium deposition as a result of measures and agreements by goverment (Cals and Roelofs, 1990)

In the short term active restoration measures may be needed to stop the deterioration of the habitats of the endangered soft water macrophytes and to counteract the effects of atmospheric deposition Moreover, after reaching the target loads in the next decade, many pools will probably not recover spontaneously and additional measures will be needed to restore the threatened or lost communities, as shown by mesocosm experiments (Brouwer *et al* , 1996)

Based upon previous investigations (Bellemakers *et al* , 1990, Bellemakers *et al* , 1993), the surface waters studied have been divided into different types according to the following criteria (i) their former and present water chemistry (particularly pH and buffering capacity), (ii) their vegetation and (iii) the cause of their deterioration The following types are distinguished

Type 1 originally weakly acid, oligotrophic moorland pools, acidified by atmospheric deposition Mean pH 4-5, alkalinity 0 0-0 2 meq l^{-1}, phytosociological alliance Sphagno-Utricularion

Type 2: originally neutral to weakly buffered, oligotrophic moorland pools, acidified by atmospheric deposition in a later stage. Mean pH 5-6; alkalinity 0.1-0.5 meq l^{-1}; phytosociological alliance *Littorellion*, in particular the Isoëto-Lobelietum association.

Type 3: weakly to moderately buffered moorland pools, first eutrophicated, then acidified by atmospheric deposition. Mean pH 6-7; alkalinity 0.1-1.0 meq l^{-1}; phytosociological alliance Hydrocotylo-Baldellion

Type 3A: Surface waters, first eutrophicated, then acidified by atmospheric deposition

Type 3B: Surface waters, eutrophicated by atmospheric deposition and inflow of surface water rich in nutrients

The whole-lake experiments of this study are based on several laboratory and enclosure experiments (Bellemakers and Van Dam, 1992; Maessen *et al.*, 1992; Bellemakers *et al.*, 1994). The aim of the study was to investigate the possibilities for recovery of acidified and/or eutrophicated shallow surface waters and moorland pools.

Materials and Methods

Study sites and the restoration measures applied

All the pools studied were acidified during recent decades, except for the Beuven. Some morphometric characteristics are presented in Table 1.

The heathland pools of the Tongerense Heide (Type 1; Figure 1) have perched water tables and their catchment areas are hardly larger than their surface areas. They are exclusively fed by rain water and thus acidified by atmosperic deposition (Heil and Diemont, 1983). More details on these

Table 1: Morphometric characteristics of the investigated pools and the applied restoration measures.

Water	location	surface area (10^3 m^2)	mean depth (cm)	removal of sapropelium	kind of pollution	restoration measure(s)	Type (see text)
Tongerense Heide	52°20'N 5°55'E	≈1	30	no	acidification	liming	1
Ven bij Schaijk	51°45'N 5°35'E	7	75	no	acidification	liming	2
Padvindersven	50°32'N 4°38'E	25	60	yes	ac. & eutr.	removal liming	3A
Beuven	51°24'N 5°39'E	750	50	yes	eutrophication	removal inlet water	3B

pools are presented by Bellemakers and Van Dam (1992). The fungal infection rate of the eggs of the moor frog (*Rana arvalis* Nilsson) in the pools has increased over the last decades (Leuven, 1988). Eight pools, where egg deposition of moor frogs had been observed in previous years, were

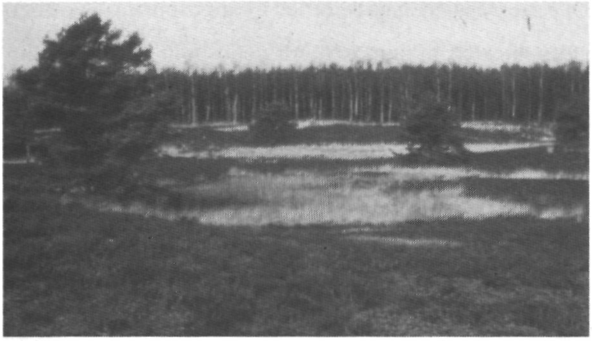


Figure 1: A heathland pool of the Tongerense Heide (J. van Osch, IBN/DLO, 1988).

Figure 2: Liming a heathland pool of the Tongerense Heide (J. van Osch, IBN/DLO, 1988).

Figure 3: Overview of the Ven bij Schaijk (J. van Osch, IBN/DLO, 1988).

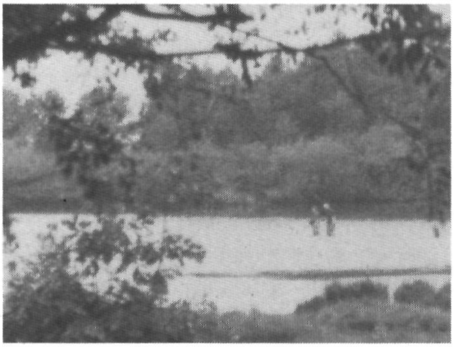


Figure 4: Overview of the Padvindersven (M. Maessen, 1989).

selected. Four of them were limed during, 1988 and 1989 (marlstone, grain size < 3 mm; 0.1 kg m^{-3}; Figure 2), whereas the other four pools were used as untreated controls.

The Ven bij Schaijk (Type 2; Figure 3) is rather flat without much vegetation development. This pool was oligotrophic and weakly buffered, before the recent acidification. In 1975 *Luronium natans* (L.) Rafin. dominated the Ven bij Schaijk (pers. comm. H. van de Steeg). Until 1984, specimens of this macrophyte were found (pers. comm. C. den Hartog). At least since 1979, the south-eastern part of the moorland pool has been dominated by *J. bulbosus* and *Sphagnum* spec. The pool was limed in December 1987 (500 kg fresh marlstone; large pieces, grain size maximum 10 cm Ø), in August 1988 (marlstone, grain size < 3 mm; same dose) and finally in June 1989 (marlstone, grain size < 3 mm; 1000 kg).

The Padvindersven (Type 3A; Figure 4) was originally eutrophicated by the supply of nutrient-rich, calcareous ground water and later on the pool became acidified by atmospheric deposition. The bottom became covered with a sapropelium layer with a fine gyttja-like structure. Until 1957, *Littorella uniflora* was abundant in this pool (Arts, 1990). In 1987, the pool was overgrown with *Drepanocladus fluitans* (Hedw.) Warnst. and *J. bulbosus*. The sapropelium layer was removed in autumn 1989 (Figure 5). Thereafter, the pool was treated annually in spring with limestone (marlstone, grain size < 3 mm; 0.1 kg m^{-3}) to counteract re-acidification.

Figure 5: Restoration of the Padvindersven (M. Maessen, 1989).

Figure 6: Restoration of the Beuven (R. Buskens, Grontmij nv., 1985).

The Beuven (Type 3B) has originally been eutrophicated by the inflow of nutrient-rich brook water. This moorland pool was distinguished by the presence of a well-developed community of soft water macrophytes (Isoëto-Lobelietum) with many rare species such as *Littorella uniflora*, *Lobelia dortmanna*, *Luronium natans* and *Isoetes echinospora* Durieu. During the period of eutrophication, a thick sapropelium layer was formed and the Beuven became almost overgrown by *Phragmites australis* (Cav.) Trin. ex Steud. The original plant communities, dominated by isoetid soft water macrophytes strongly declined and almost disappeared. To create a stable water chemistry and appropriate germination conditions the sapropelium layer including *Phragmites* was removed in winter 1985-1986 (Figure 6). To prevent the Beuven from acidification, buffered brook water has been supplied. In order to remove nitrogen and phosphorus before it could flow into the Beuven the brook water was first directed into a shallow basin south of the Beuven (residence time of 3 weeks;

Buskens, 1989).

Water sampling and vegetation monitoring

Water samples for chemical analysis were taken before and after the restoration measures in the centre of the pools or in the case of Beuven close to the water inflow. All samples were taken at least every three months. The samples were transported directly to the laboratory in closed polyethylene airtight dark bottles (2 l).

The vegetation of the Ven bij Schaijk and the Padvindersven was characterized by relevees by a modified Braun-Blanquet approach and the phytosociological communities were named according to Westhoff and Den Held (1969) and Schaminée *et al.* (1992). The surface areas of vegetation types dominated by *Sphagnum* spec., *J. bulbosus*, *D. fluitans* and soft water macrophytes were mapped and calculated.

The presence of macrophytes in the Beuven was recorded from 1936 to 1993 in time intervals of 10 years, based on data from literature and reports from various researchers (Buskens, 1989).

Field and laboratory measurements

Measurements of pH, alkalinity and turbidity were carried out within one day after sampling. pH was measured with a GK2501B combined pH electrode, connected to a Radiometer Copenhagen PHM82 pH/mV meter. Alkalinity was determined by titrating 100 ml of water with 0.01 N HCl down to pH 4.2 (Stumm and Morgan, 1981). A subsample was passed through a Whatman GF/C filter (1.2 μm), stored in iodated polyethylene bottles, and frozen at -28 °C until chemical analysis. Colorimetric measurements with a continuous flow auto-analyser were conducted for nitrite/nitrate according to Kamphake *et al.* (1967), total ammonia according to Kempers and Zweers (1986) and ortho-phosphate according to Henriksen (1965). Determination of sodium and potassium was carried out flame-photometrically; calcium, magnesium and aluminum were determined with an Inductively Coupled Plasma spectrophotometer (ICP) type IL Plasma 200. Sulfate concentrations were determined gravimetrically, according to Technicon Auto-analyzer Methodology (1981) and the turbidity was determined by means of a Toho Dentan model FN5 turbiditymeter.

Statistics

To calculate the geometric means and the variances of the chemical data before and after the restoration measures, a log-transformation was used, to obtain normality. The measurements within a year after the introduction of the control measures were deleted from the calculations of the geometric means to avoid the disturbances as a consequence of removal of the organic sapropelium layer. These geometric means were compared with the T-test (Sokal and Rohlf, 1981).

The development in water chemistry after the treatments was plotted in time (time trends) and the differences were tested by using the signed-rank Wilcoxon test (Sokal and Rohlf, 1981).

Results

Tongerense Heide

As shown in Figure 7 and Table 2 the pH and alkalinity increased ($p < 0.01$) after liming. In the limed pools the pH increased from 4 to 5 and the alkalinity reached maximum values of 300 μeq l^{-1}. Significant changes in ortho-phosphate and in ammonium/nitrate ratio were not found. Except for pH, alkalinity and calcium concentrations, the water chemistry parameters did not show significant differences between the limed and control pools (Table 2). In the two years after liming no changes in vegetation composition were observed.

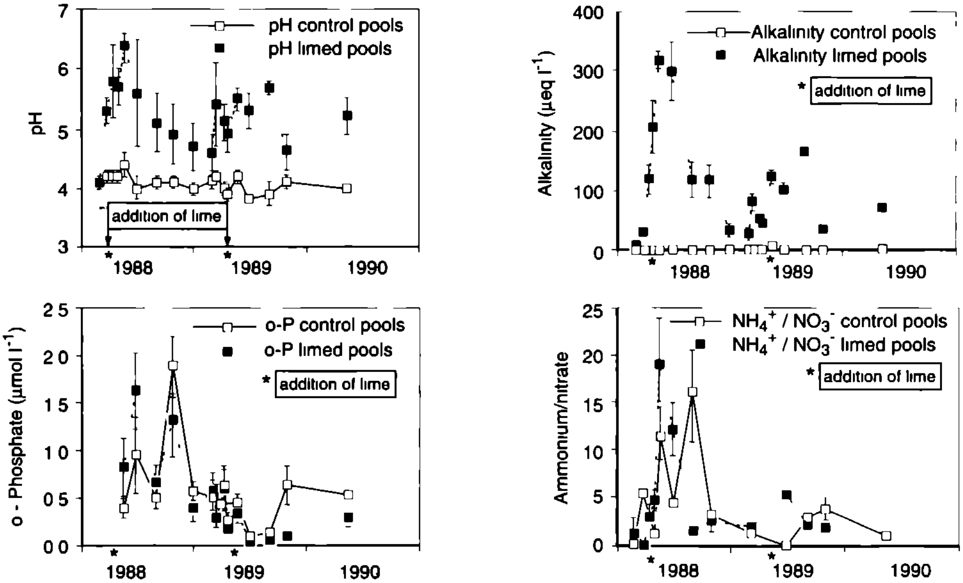


Figure 7: The pH, alkalinity, ortho-phosphate concentrations and the ammonium/nitrate molar ratio in 4 limed and 4 control heathland pools at the Tongerense Heide

Ven bij Schaijk

After liming in December 1987, the pH and alkalinity increased slightly and then rapidly decreased within a few months. The second liming resulted in a more obvious increase and the third liming experiment showed a significant increase in pH and alkalinity, up to 8 and 500 μeq l^{-1}, respectively (Figure 8). In the years after these lime additions (1990 to 1992), the pH and alkalinity of the water layer returned to pre-treatment levels. The ortho-phosphate concentrations showed little difference

Table 2: Geometric means with minimum and maximum values of selected chemical variables of the Tongerense Heide before and after restoration measures.

parameter	before restoration measures		after restoration measures		
	mean	(min-max)	mean	(min-max)	
Number of observations	84		86		
pH	4.1	(3.6-4.7)	5.1	(3.7-6.9)	***
Alkalinity (μeq l^{-1})	1	(0-32)	88	(0-454)	***
Ammonium (μmol l^{-1})	25	(0-800)	16	(0-230)	
Nitrate (μmol l^{-1})	7	(0-90)	6	(0-76)	
o-Phosphate (μmol l^{-1})	0.70	(0.00-2.10)	0.46	(0.00-5.50)	
Aluminum (μmol l^{-1})	6	(0-126)	4	(0-97)	
Calcium (μmol l^{-1})	41	(8-424)	97	(10-743)	***
Magnesium (μmol l^{-1})	28	(7-261)	30	(11-318)	
Sulphate (μmol l^{-1})	64	(3-1595)	37	(6-1091)	
Chloride (μmol l^{-1})	64	(14-1514)	60	(10-2205)	
Seston (mg l^{-1})	11.6	(0.2-55.8)	10.4	(1.0-88.8)	
Turbidity (ppm $PtCl_2$)	4.3	(1.0-9.0)	4.7	(1.0-13.0)	

***: Significant change: $p < 0.01$ (T-test)

after liming. The ammonium/nitrate ratio stabilized at a lower level after liming ($p < 0.05$). When all data are divided into two groups (before December 1987 and after December 1988) pH, alkalinity and calcium concentrations of the water layer increased significantly after liming (Table 3). Figure 9 shows the decrease in surface area covered by *Sphagnum* spec. (S) and the increase in surface area covered by *J. bulbosus* (Js) after liming of the water layer. The floristic composition did not change markedly, and no endangered soft water macrophytes returned. In contrast, there was an expansion of *J. bulbosus*.

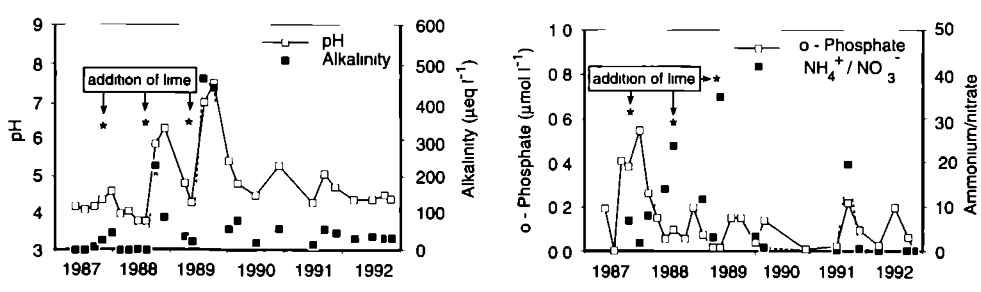


Figure 8: The pH, alkalinity, ortho-phosphate concentrations and the ammonium/nitrate molar ratio in the Ven bij Schaijk.

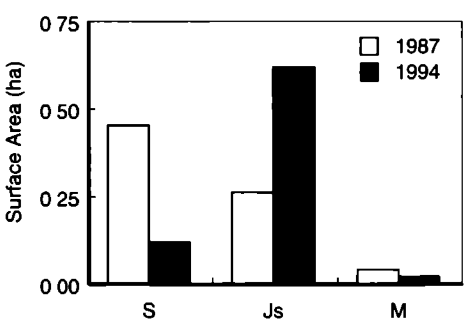
Table 3 Geometric means with minimum and maximum values of selected chemical variables of the Ven bij Schaijk before and after restoration measures

parameter	before restoration measures		after restoration measures		
	mean	(min-max)	mean	(min-max)	
Number of observations	25		39		
pH	4.3	(4.1-4.6)	4.9	(4.5-5.7)	***
Alkalinity (µeq l^{-1})	12	(0-62)	85	(0-580)	***
Ammonium (µmol l^{-1})	27	(6-92)	13	(3-43)	*
Nitrate (µmol l^{-1})	1	(0-1)	9	(2-54)	***
o-Phosphate (µmol l^{-1})	0.18	(0-0.41)	0.10	(0.02-0.13)	**
Aluminum (µmol l^{-1})	45	(30-57)	21	(7-70)	**
Calcium (µmol l^{-1})	162	(129-175)	269	(140-391)	***
Magnesium (µmol l^{-1})	172	(39-198)	194	(111-337)	
Sulphate (µmol l^{-1})	521	(396-610)	523	(78-953)	
Chloride (µmol l^{-1})	605	(337-724)	600	(317-782)	
Seston (mg l^{-1})	3.8	(1.0-9.0)	3.7	(0.1-31.2)	
Turbidity (ppm $PtCl_2$)	3.4	(0.0-9.0)	3.0	(0.0-31.0)	*

* Significant change $p < 0.1$, ** Significant change $p < 0.05$, *** Significant change $p < 0.01$ (T-test)

Padvindersven

Because of a drought period during 1989 and 1990, the number of samples after removal of the sapropelium layer was relatively low. After the lime addition, the pH and alkalinity stabilized within one year at a level somewhat higher than before the treatments (Figure 10, 5 and 50 µeq l^{-1}, respectively). Prior to the removal of the organic sapropelium layer, ammonium and ortho-phosphate fluctuated strongly: the maximum concentrations could be very high, up to 1000 µmol l^{-1} and 0.68 µmol l^{-1}, respectively (Figure 4 and Table 4). After the sapropelium removal the ammonium and ortho-phosphate concentrations were significantly lower and stabilized, whereas the nitrate and calcium concentrations increased significantly (Table 4).



S — dominance of *Sphagnum* spec.
Js — dominance of *Juncus bulbosus*, local *Sphagnum* spec.
M — dominance of *Molinia caerulea*

Figure 9 Surface area dominated by different vegetation types in the Ven bij Schaijk (1987 and 1994)

Table 4: Geometric means with minimum and maximum values of selected chemical variables of the Padvindersven before and after restoration measures.

parameter	before restoration measures		after restoration measures		
	mean	(min-max)	mean	(min-max)	
Number of observations	37		7		
pH	4.6	(3.7-5.5)	5.1	(4.7-5.7)	**
Alkalinity (μeq l^{-1})	42	(0-158)	39	(0-68)	
Ammonium (μmol l^{-1})	111	(0-1000)	20	(4-71)	*
Nitrate (μmol l^{-1})	6	(0-21)	46	(10-116)	***
o-Phosphate (μmol l^{-1})	0.34	(0.12-0.68)	0.08	(0.04-0.12)	*
Aluminum (μmol l^{-1})	11	(1-72)	19	(7-70)	
Calcium (μmol l^{-1})	101	(34-449)	258	(140-391)	***
Magnesium (μmol l^{-1})	56	(29-95)	85	(31-183)	
Sulphate (μmol l^{-1})	262	(0-1012)	296	(34-572)	
Chloride (μmol l^{-1})	312	(85-1051)	367	(216-637)	
Seston (mg l^{-1})	12.4	(0.0-76.0)	5.6	(0.0-19.1)	
Turbidity (ppm $PtCl_2$)	-		7	(1-24)	

*: Significant change: $p < 0.1$; **: Significant change: $p < 0.05$; ***: Significant change: $p < 0.01$ (T-test)

Before the restoration, the Padvindersven contained a plant community dominated by *J. bulbosus* and *D. fluitans* (Figure 11, JD). In autumn 1989 the sapropelium removal was performed and since then the area, characterized by the presence of soft water species (L; *Littorella uniflora*, *Ranunculus ololeucos* Lloyd, *Lythrum portula* (L.) D.A. Webb and *Hypericum elodes* L.) increased (Figure 11). However, after 1993 these species were partly outcompeted by *J. bulbosus*.

Beuven

Prior to sapropelium removal and the reduction in nutrient inputs in the winter of 1985, pH and alkalinity were high (pH: 7-10; alkalinity: 0.7-1.4 meq l^{-1}). After the control measures, pH and alkalinity stabilized at lower levels (5-6 and 300 μeq l^{-1}, respectively). The concentrations of ortho-

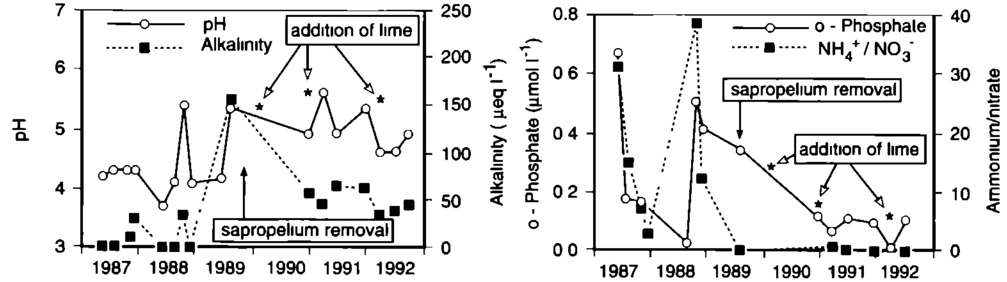
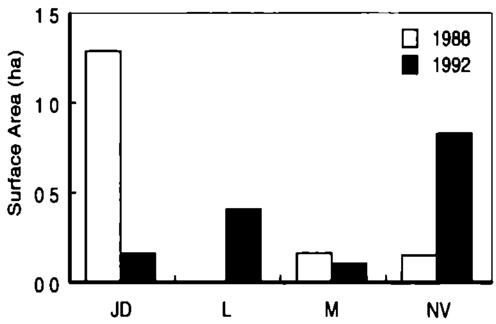


Figure 10: The pH, alkalinity, ortho-phosphate concentrations and the ammonium/nitrate molar ratio in the Padvindersven.



JD dominance of *Juncus bulbosus* and *Drepanocladus fluitans*
L presence of vegetation with soft water macrophytes (*Littorella uniflora*, *Hypericum elodes*, *Ranunculus ololeucos* and *Lythrum portula*)
M dominance of *Molinia caerulea*
NV no vegetation, sandy soil

Figure 11 Surface area dominated by different vegetation types in the Padvindersven (1988 and 1992)

phosphate and the ammonium/nitrate ratio fluctuated very strongly before the winter of 1985, and were incidentally extremely high. After the winter of 1985 both parameters stabilized at low levels (Figure 12).

Prior to removal of the sapropelium layer, the average nutrient concentrations were high, whereas after the restoration measures they all decreased significantly, except for nitrate (Table 5). Due to the restoration measures (sapropelium removal and controlled inflow of buffered water after reduction in nutrient inputs), no acidification of the water layer (mean alkalinity: 97 $\mu eq\ l^{-1}$) was observed. The mean turbidity decreased strongly after the restoration.

The presence of aquatic macrophytes in the Beuven has been recorded since 1936 (Table 6). Only very few species disappeared during the period 1936-1985, but their abundance became very low

Table 5: Geometric means with minimum and maximum values of selected chemical variables of the Beuven before and after restoration measures

parameter	before restoration measures		after restoration measures		
	mean	(min-max)	mean	(min-max)	
Number of observations	80		70		
pH	6.9	(4.6-10.4)	5.7	(4.7-7.9)	***
Alkalinity ($\mu eq\ l^{-1}$)	498	(2-2230)	97	(0-705)	***
Ammonium ($\mu mol\ l^{-1}$)	32	(6-92)	14	(1-75)	***
Nitrate ($\mu mol\ l^{-1}$)	34	(0-68)	35	(0-45)	
o-Phosphate ($\mu mol\ l^{-1}$)	1.12	(0.70-1.50)	0.38	(0.05-1.75)	***
Aluminum ($\mu mol\ l^{-1}$)	19	(3-75)	6	(1-26)	***
Calcium ($\mu mol\ l^{-1}$)	647	(148-1225)	378	(143-773)	***
Magnesium ($\mu mol\ l^{-1}$)	244	(45-480)	132	(38-227)	***
Sulphate ($\mu mol\ l^{-1}$)	696	(83-1406)	596	(162-693)	
Chloride ($\mu mol\ l^{-1}$)	838	(270-1020)	531	(178-802)	***
Turbidity (ppm $PtCl_2$)	41.0	(12.0-49.0)	3.5	(0.0-8.6)	***

***: Significant change $p < 0.01$ (T-test)

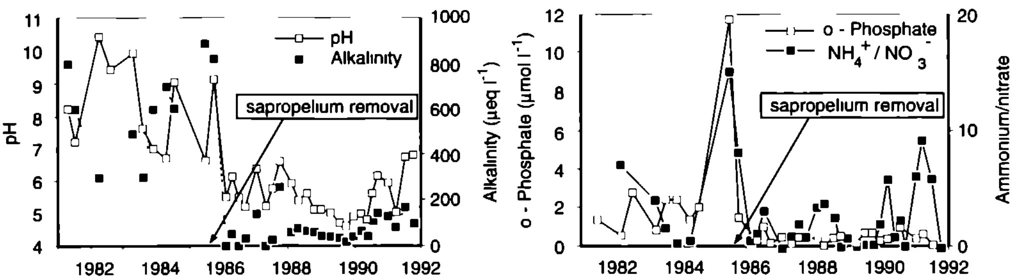


Figure 12: The pH, alkalinity, ortho-phosphate concentrations and the ammonium/nitrate molar ratio in the Beuven.

After the removal of the sapropelium layer almost all species returned or increased in numbers. Many characteristic soft water species became very abundant, especially after the droughts of 1989 and 1990.

Table 6: The presence of aquatic species and endangered soft water species (underlined) in the Beuven from 1936 to now.

period:	1	2	3	4	5	6
Potamogeton gramineus L.						*
<u>*Potamogeton obtusifolius* Mert. & Koch</u>						*
Sparganium angustifolium Michx.	*					*
<u>*Apium inundatum* (L.) Rchb. f.</u>				*		*
Deschampsia setacea (Hudson) Hackel				*		*
Pilularia globulifera L.	*	*	*			*
<u>*Isoetes echinospora* Durieu</u>	*	*	*	*		*
Sparganium minimum Wallr.		*	*	*	*	*
<u>*Littorella uniflora* (L.) Aschers.</u>	*	*	*	*	*	*
<u>*Lobelia dortmanna* L.</u>	*	*	*	*	*	*
<u>*Echinodorus repens* (Lamk.) Kern & Reichgelt</u>	*	*	*	*	*	*
<u>*Luronium natans* (L.) Rafin.</u>	*	*	*	*	*	*
Hypericum elodes L.	*	*	*	*	*	*
Juncus bulbosus L.	*	*	*	*	*	*
<u>*Eleocharis multicaulis* (J.E. Smith) J.E. Smith</u>	*	*	*	*	*	*
<u>*Elatine hexandra* (Lapierre) DC.</u>	*		*	*	*	*
Eleocharis acicularis (L.) Roemer & Schultes	*		*	*	*	*
Potamogeton pusillus L.	*			*	*	*
Lythrum portula (L.) D.A. Webb		*	*	*	*	*
Potamogeton natans L.		*	*	*	*	*
Lemna minor L.			*	*	*	*
Utricularia australis R.Br.				*	*	*
Callitriche spec.				*	*	*
Polygonum amphibium L.				*	*	*
Riccia fluitans L.				*	*	*
Lemna trisulca L.				*		*

period 1: 1936 - 1945; period 2: 1946 - 1955; period 3: 1956 - 1965; period 4: 1966 - 1975; period 5 1976 - 1985 and period 6: 1986 - now '*' = present (after Buskens, 1989)

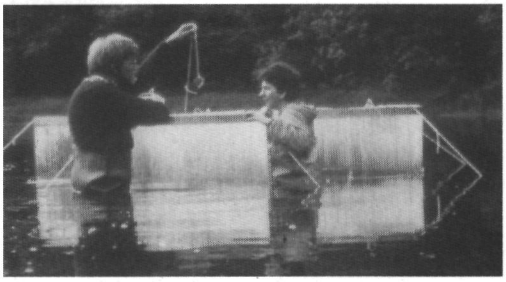


Figure 13: The enclosure experiment of the Ven bij Schaijk (J. van Osch, IBN/DLO, 1988).

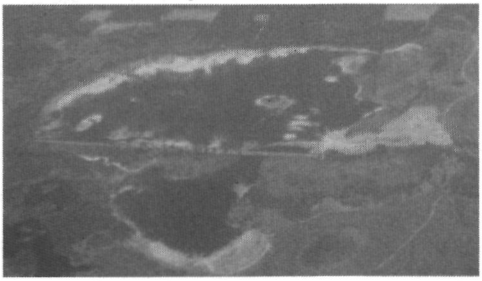


Figure 14: Overview of the Beuven after removal of the sapropelium layer. At the side of the Beuven one can clearly see the sandy banks (air photograph, 1986).

Discussion

Water chemistry

In acidified shallow pools with a thin sapropelium layer (Types 1 and 2), the water chemistry hardly showed any changes after subtle liming of the water layer, particularly when they had been dried out. Part of the added calcium ions were exchanged against the protons stored in the sediment absorption complex, causing an accelerated re-acidification (Ripl and Lindmark, 1979; Lindmark, 1982; Lindmark, 1984).

The same processes took place in enclosure experiments in the Ven bij Schaijk after liming the water layer (Bellemakers *et al.*, 1994; Figure 13). This re-acidification also led to elevated aluminum levels in the water layer. Liming projects in large lakes in the U.S.A. led to low aluminum concentrations in the water layer. In the following years also re-acidification took place and led to increased levels of aluminum levels in the water layer (Schofield *et al.*, 1986).

After liming pH and alkalinity initially increased, but soon decreased to pre-treatment levels, due to re-acidification by high atmospheric loads (Schuurkes *et al.*, 1988), and/or cation exchange in the sediment.

From enclosure experiments conducted earlier (Bellemakers *et al.*, 1994) and the experiments described in this paper, it appeared that the nutrient concentrations of the water layer did not increase in limed pools without removal of the sapropelium layer, although, some eutraphentic and saprophilous diatoms were found in the water layer of the pools of the Tongerense Heide and the Ven bij Schaijk (Bellemakers and Van Dam, 1992). Probably, internal eutrophication took place, at low level, by mineralization of the thin sapropelium layer (Schindler, 1986; Curtis, 1989; Roelofs, 1991).

In order to avoid eutrophication, the sapropelium in the Padvindersven (Type 3A) was removed prior to liming. After performance of the sapropelium removal and liming, the water layer of this moorland pool became weakly buffered and poor in nutrients. The slightly increased pH and alkalinity stimulate the nitrification rate leading to decreased ammonium and increased nitrate levels and thus to decreased ammonium/nitrate ratios (Schuurkes *et al.*, 1986). The ortho-phosphate concentrations remained at low levels after sapropelium removal and liming. The low concentrations of inorganic carbon, nitrogen and phosphate led to clear water with only slight algal growth.

The drought period of 1989 and 1990

In the dry years 1989 and 1990 the Padvindersven almost dried out, leading to aeration of the sediments. This may have had severe consequences for water chemistry (Van Dam, 1988). The internal generation of alkalinity as a consequence of sulphate reduction in the acidified sediments decreased. These processes have been previously described by Kelly *et al.* (1987) and Kelly (1988). Besides, drying out of surface waters by fluctuating water tables may also improve the aerobic condition of the sediment. This aeration stimulates the germination and development of soft water macrophytes (Arts, 1990).

Excessive growth of Juncus bulbosus

After liming, re-acidification and internal eutrophication of the water layer occurred, accompanied by high levels of carbon dioxide, leading to a mass development of *Juncus bulbosus*, particularly in the Ven bij Schaijk. This process was earlier described by Roelofs (1983) after acidification of moorland pools. Culture experiments and seed bank experiments also showed a luxurious growth of *Juncus bulbosus* after liming of the water layer, probably due to an accelerated decomposition rate of the organic sapropelium layer (Bellemakers *et al.*, 1996). Because of this dominance of *J. bulbosus* the diversity of soft-water macrophytes decreased and because of this highly undesirable development, these waters may not be restored without removal of the organic nutrient-rich sapropelium layer.

The removal of the sapropelium layer and the liming had at first a positive impact on the vegetation development of the Padvindersven. Already in 1990 there was a spectacular development of soft water macrophyte communities. The diversity of the plant communities increased and became similar to the situation before acidification and eutrophication (Van der Voo, 1957; Arts, 1990).

Recently, however, excessive *J. bulbosus* growth was observed, although the organic sapropelium layer had been removed. Expansive growth of *J. bulbosus*, 3-5 years after the first lime applications has also been noticed in limed Norwegian lakes without thick sapropelium layers (Roelofs *et al.*, 1994). Therefore, regulated inflow of buffering substances can lead to more desirable developments of the water chemistry and thus also the vegetation.

Inlet of brook water after removal of the sapropelium layer

The Beuven was originally eutrophicated by inflow of nutrient rich brook water, but was seriously threatened by acidification after restoration because of the removal of the buffering sapropelium layer (Figure 14). To avoid acidification, eutrophication and massive expansion of *J. bulbosus*, alkaline brook water, was used to buffer the water of the Beuven, after removal of nutrients in a shallow basin. This measure has functioned well up to now (Buskens, 1989; Buskens, 1994), as the degree of buffering could be regulated exactly. The pH and alkalinity of the sediment pore water could be kept at low level and decomposition processes also remain at low levels (50 - 200 μeq l^{-1}; Kok and Van de Laar, 1991). This may be the reason that no mass development of *J. bulbosus* in the Beuven has occurred until now (Roelofs *et al.*, 1994).

The increased pH and alkalinity in the first year after the restoration measures were caused by disturbance of the environment, as a consequence of removing the organic sapropelium layer, but after 1986 pH and alkalinity returned to their original levels. The nutrient concentrations also decreased after the removal of the sapropelium layer and the inlet of water from the shallow basin. In winter, when there is less microbiological activity, enhanced nitrogen concentrations, comparable with the quantity of nitrogen in atmospheric deposition were found (Schuurkes *et al.*, 1986). After the control measures (sapropelium removal and inflow of buffered water after reduction in nutrient inputs), the main characteristics of the original water chemistry returned: the Beuven again became a weakly buffered, relatively oligotrophic moorland pool.

In the first year after restoration, germination and establishment of soft water species was observed. Later on, all species, observed in earlier times, returned in large numbers.

Conclusions

The shallow surface waters studied in the Netherlands show remarkable differences in geomorphology, type of sediment, hydrology, water chemistry and vegetation. They may be divided into four categories according to their acidification and eutrophication history. Liming of the water layer without removal of the sapropelium layer in Type 1 and 2 waters resulted in acceptable water chemistry. However, the endangered plant species have not returned (Type 2), as a consequence of reductive processes in the sediment. It is concluded that the sapropelium layer produced protons after liming, which caused an accelerated re-acidification. Liming of the water layer without removing the organic material resulted in internal eutrophication processes, despite the low concentrations of nutrients in the water layer. These nutrients may be recycled by phytoplankton or diatoms (Birch and Spyridakis, 1981; Van Dam *et al.*, 1988).

Long-term biomonitoring shows that direct liming of weakly buffered waters to counteract acidification is not an effective treatment in the long run After positive results for water chemistry and plant growth in the first years, liming caused a massive expansion of *Juncus bulbosus* within the following years and these observations have been confirmed during liming experiments in Norway (Roelofs *et al* , 1994)

Creating inflows of buffered water seems favourable as a measure against acidification The amount of the inflow water to be diverted can be calculated exactly, in contrast to liming (because of the low solubility of lime in water and the accumulation of the undissolved lime in the sediments) After inflow of buffered water, no increase of pH and alkalinity of the sediment pore water occurs and no mass invasion of *J bulbosus* has been observed

The final goal of this kind of management techniques in shallow surface waters was to develop a water chemistry, comparable to the natural shallow, weakly buffered, oligotrophic surface water chemistry before the impact of cultural acidification and eutrophication processes (pH above 5, alkalinity between 50 and 1000 $\mu eq\ l^{-1}$, low concentrations of nutrients and low ammonium/nitrate ratios) By creating these circumstances, endangered soft water macrophytes can return and the natural values of these surface waters can be restored (Arts, 1990) Since introduction of such restoration measures, the number of locations where endangered soft water species can occur, has been significantly increased in the Netherlands

Acknowledgements

The authors are indebted to Dr H van Dam, Prof Dr C den Hartog and Drs E Brouwer for critically reading and improving the manuscript We also thank Hanske van Westreenen and the late Jan Peters from the illustrations department of the Faculty of Science for preparing the drawings and Mrs G M Verheggen-Kleinheerenbrink for technical assistance during chemical analysis We also thank the students P W M van Beers, M B G Koopmans, G H S Kurstjens and R H M Nabben for practical assistance in the field and experimental work This study was financed by the Ministry of Agriculture, Nature Conservation and Fisheries

References

Arts, G.H.P., 1990. Deterioration of atlantic soft-water systems and their flora, a historical account. Thesis University of Nijmegen, 197 pp.

Arts, G.H.P., Van der Velde, G., Roelofs, J.G.M. and Van Swaay, C.A.M., 1990. Successional changes in the soft-water macrophyte vegetation of (sub)atlantic, sandy, lowland regions during this century. Freshwat. Biol. 24: 287-294.

Asman, W.A.H., 1987. Atmospheric behaviour of ammonia and ammonium. Thesis, Agricultural University of Wageningen, 173 pp.

Bellemakers, M.J.S., Maessen M. and Verheggen, G.M., 1990. Restauratie van verzuurde en geëutrofieerde zwak gebufferde oppervlaktewateren; mogelijkheden tot herstel. Department of Aquatic Ecology and Biogeology, University of of Nijmegen, by order of the Ministry of Housing, Physical Planning and Environment, 98 pp. (in Dutch).

Bellemakers, M.J.S. and Van Dam, H., 1992. Improvement of breeding success of the moor frog (Rana *arvalis*) by liming of acid moorland pools and the consequences of liming for water chemistry and diatoms. Environm. Pollut. 78: 165-171.

Bellemakers, M.J.S., Maessen, M., Cals, M.J.R. and Roelofs, J.G.M., 1993. Effectgerichte maatregelen tegen verzuring en eutrofiëring van oppervlaktewateren. Report monitoringsprogramma eerste fase. Department Ecology, Section Environmental Biology University of of Nijmegen, by order of the Ministry of Agriculture, Nature Conservation and Fisheries. 148 pp. (in Dutch)

Bellemakers, M.J.S., Maessen, M. and Roelofs, J.G.M., 1994. Effects of liming on water chemistry in shallow acidified pools in the Netherlands: enclosure experiments. Water, Air, Soil Pollut. 73: 131-142.

Bellemakers, M.J.S., Maessen, M., Verheggen, G.M. and Roelofs, J.G.M., 1996. Effects of liming on water chemistry in shallow acidified moorland pools: the germination and development of aquatic macrophytes. Aquat. Bot. 54: 37-50.

Bobbink, R., Boxman, D., Fremstad, E., Heil, G., Houdijk, A. and Roelofs, J., 1992. Critical loads for nitrogen eutrophication of terrestrial and wetland ecosystems based upon changes in vegetation and fauna. In: Grennfelt, P. and Thörnelöf, E. (eds.): Critical loads for nitrogen, p 111-159. Nord (miljorapport) 1992: 41, Nordic Council of Ministers, Copenhagen.

Brouwer, E., Bobbink, R., Meeuwsen, F. and Roelofs, J.G.M., 1996. Recovery from acidification in aquatic mesocosms after reducing ammonium and sulphate deposition. Aquat. Bot. 56: 119-130.

Buskens, R.F.M., 1989. Beuven: herstel van een oecosysteem. Report Department of Aquatic Ecology and Biogeology, University of Nijmegen, by order of the Ministry of Agriculture, Nature Conservation and Fisheries. pp. 154 (in Dutch).

Buskens, R.F.M., 1994. Beuven blijvend hersteld? De Levende Natuur 95: 211-217 (in Dutch).

Birch, P.B. and Spyridakis, D.E., 1981. Nitrogen and phosphorus recycling in Lake Sammamish, a temperate mesotrophic lake. Hydrobiologia 80: 129-138.

Cals, M.J.R. and Roelofs, J.G.M., 1990. Prae-advies effectgerichte maatregelen tegen verzuring en eutrofiëring in oppervlaktewateren. Report Department of Aquatic Ecology and Biogeology, University of of Nijmegen, by order of the Ministry of Agriculture, Nature Conservation and Fisheries, 916 pp. (in Dutch).

Curtis, P.J., 1989. Effects of hydrogen ion and sulfate on the phosphorus cycle of a Precambrian Shield lake. Nature 337: 156-158.

Den Hartog, C., 1986. The effects of acid and ammonium deposition on aquatic vegetations in The Netherlands. Proc. 1st. Internat. Symp. Watermilfoil (*Myriophyllum spicatum*) and related Haloragaceae species, 1985, Vancouver, B.C., p. 51-58.

Den Hartog, C. and Van der Velde, G., 1988. Structural aspects of aquatic plant communities. In: J.J. Symoens (ed.). Vegetation of inland waters. Handbook of vegetation science 15: 113-153.

Drabløs, D. and Tollan, A., (eds.), 1980. Ecological impact of acid precipitation. Proceedings International Conference of Ecological Impact of Acid Precipitation, Norway 1980, SNSF-project, Oslo-Aas 1980.

Henriksen, A., 1965. An automated method for determining low-level concentrations of phosphate in fresh and saline waters. Analyst (London) 90: 29-34.

Heil, G.W. and Diemont, W.H., 1983. Raised nutrients levels change heathland into grassland. Vegetatio 53: 113-120.

Kamphake, L.H., Hannah, S.A. and Cohen, J.M., 1967. Automated analysis for nitrate by hydrazine reduction. Water Res. 1: 206.

Keeley, J.E., 1983. Dark CO_2-fixation and diurnal malic acid fluctuations in the submerged aquatic *Isoetes storkii*. Oecologia 48: 322-333.

Kelly, C.A., Rudd, J.W.M., Hesslein, R.H., Schindler, D.W., Dillon, P.J., Driscoll, C.T. and Gherini, S., 1987. Prediction of biological acid neutralisation in acid-sensitive lakes. Biogeochemistry 3: 129-140.

Kelly, C.A., 1988. Toward improving comparisons of alkalinity generation in lake basins. Limnol. Oceanogr. 33: 1635-1637.

Kempers, A.J. and Zweers, A., 1986. Ammonium determination in soil extracts by the salicylate method. Commun. Soil Sci. Plant Anal. 17: 715-723.

Kok, C.J. and Van de Laar, B.J., 1991. Influence of pH and buffering capacity on the decomposition of *Nymphaea alba* L. detritus in laboratory experiments: a possible explanation for the inhibition of decomposition at low alkalinity. Verh. Internat. Verein. Limnol. 24: 2689-2692.

Leuven, R.S.E.W., 1988. Impact of acidification on aquatic ecosystems in the Netherlands with emphasis on structural and functional changes. Thesis University of Nijmegen, 181 pp.

Leuven, R.S.E.W., Van der Velde, G. and Kersten, H.L.M., 1989. Interrelations between pH and other physico-chemical factors of Dutch soft waters. Arch. Hydrobiol. 126: 27-51.

Lindmark, G.K., 1982. Acidified lakes: ecosystem response following sediment treatment with sodium carbonate. Verh. Internat. Verein. Limnol. 22: 772-779.

Lindmark, G.K. 1984., Acidified lakes: sediment treatment with sodium carbonate - a remedy? Hydrobiologia. 92: 537-547.

Madsen, T.V., 1985. A community of submerged aquatic CAM-plants in Lake Kalgaard, Denmark. Aquat. Bot. 23: 97-108.

Maessen, M., Roelofs, J.G.M., Bellemakers, M.J.S. and Verheggen, G.M., 1992. The effects of aluminum, aluminum/calcium ratios and pH on aquatic plants from poorly buffered environments. Aquat. Bot. 43: 115-127.

Mortimer, C.H., 1971. Chemical exchanges between sediments and water in the Great Lakes, speculations on probable regulatory mechanisms. Limnol. Oceanogr. 16: 387-404.

Overrein, L.N., Seip, H.M. and Tollan, A., (eds.) 1981. Acid precipitation - Effects on forest and fish. Final report SNSF-project 1972-1980. Oslo-As, 175 pp.

Rao, S.S. and Dutka, B.J., 1983. Influence of acid precipitation on bacterial populations in lakes. Hydrobiologia 98: 153-157.

Ripl, W. and Lindmark, G.K., 1979. The impact of algae and nutrient composition on sediment exchange dynamics. Arch. Hydrobiol. 86: 45-65.

Roelofs, J.G.M., 1983. Impact of acidification and eutrophication on macrophyte communities in soft waters in the Netherlands: I. Field observations. Aquat. Bot. 17: 139-155.

Roelofs, J.G.M., Schuurkes, J.A.A.R. and Smits, A.J.M., 1984. Impact of acidification and eutrophication on macrophyte communities in soft waters: II. Experimental studies. Aquat. Bot. 18: 389-411.

Roelofs, J.G.M., 1991. Inlet of alkaline river water into peaty lowlands: effects on water quality and *Stratiotes aloides* L. stands. Aquat. Bot. 39: 267-293.

Roelofs, J.G.M., Brandrud, T.E. and Smolders, A., 1994. Mass invasion of *Juncus bulbosus* after liming of acidified Norwegian lakes. Aquat. Bot. 48: 187-202.

Sand-Jensen, K. and Søndergaard, M., 1979. Distribution and quantitative development of aquatic macrophytes in relation to sediment characteristics in oligotrophic Lake Kalgaard, Denmark. Freshwat. Biol. 9: 1-11.

Sand-Jensen, K., Prahl, C. and Stokholm, M., 1982. Oxygen release from roots of submerged aquatic macrophytes. Oikos 38: 349-359.

Schaminée, J.H.J., Westhoff, V. and Arts, G.H.P., 1992. Die Strandlinggesellschaften (Littorelletea Br.-Bl. et Tx. 43) der Niederlande, in europäischen Rahmen gefasst. Phytocoenologia 20: 529-558.

Schindler, D.W., 1986. The significance of in-lake production of alkalinity. Water, Air, Soil Pollut. 30: 931-944.

Schofield, C.L., Gloss, S.P. and Josephson, D., 1986. Extensive evaluation of lake liming, restocking strategies, and fish population response in acidic lakes following neutralization by liming. U.S. Fish and Wildlife Service, Eastern Energy and Land Use Team. Interim Progress Rept. NEC-86/18, 117 pp.

Schoof-Van Pelt, M.M., 1973. Littorelletea, a study of the vegetation of some amphiphytic communities of western Europe. Thesis University of Nijmegen, 216 pp.

Schuurkes, J.A.A.R., Heck, I.C.C., Hesen, P.L.G.M., Leuven, R.S.E.W. and Roelofs, J.G.M., 1986. Effects of sulphuric acid and acidifying ammonium deposition on water quality and vegetation of simulated soft water ecosystems. Water, Air, Soil Pollut. 31: 267-272.

Schuurkes, J.A.A.R., 1987. Acidification of surface waters by atmospheric deposition. Thesis University of Nijmegen, 160 pp.

Schuurkes, J.A.A.R., Jansen, J. and Maessen, M., 1988. Water acidification by addition of ammonium sulfate in sediment-water columns and in natural water. Arch. Hydrobiol. 112: 495-516.

Smits, A.J.M., Laan, P., Thier, R.H. and Van der Velde, G., 1990. Root aerenchyma, oxygen leakage patterns and alcoholic fermentation ability of the roots of some nymphaeid and isoetid macrophytes in relation to the sediment type of their habitat. Aquat. Bot. 38: 3-17.

Sokal, R.R. and Rohlf, F.J., 1981. Biometry, second edition. Freeman, San Francisco, 859 pp.

Søndergaard, M., 1979. Light and dark respiration and the effect of the lacunal system on refixation of CO_2 in submerged aquatic plants. Aquat. Bot. 6: 269-283.

Søndergaard, M. and Sand-Jensen, K., 1979. Carbon uptake by leaves and roots of *Littorella uniflora* (L.) Aschers. Aquat. Bot. 6: 1-12.

Stumm, W. and Morgan J.J., 1981. Aquatic chemistry: an introduction emphasizing chemical equilibria in natural waters. 2nd ed., Wiley Interscience, New York, 780 pp.

Technicon Auto-Analyzer Methodology, 1981. Industrial Method 635-81W, New York.

Van Dam, H., Suurmond, E. and ter Braak, C.J.F., 1981. Impact of acidification on diatoms and chemistry of Dutch moorland pools. Hydrobiologia 83: 425-459.

Van Dam, H., 1988. Acidification of three moorland pools in The Netherlands by acid precipitation and extreme drought periods over seven decades. Freshwat. Biol. 20: 157-176.

Van Dam, H., Mertens, A. and Bellemakers, M.J.S., 1988. Effects of liming on attached diatoms in experimental enclosures in an acidified moorland pool (preliminary results). Proceedings Third International Conference on the Conservation and Management of Lakes "Balaton '88". September 11-17, Keszthely, Hungary.

Van der Voo, E.E., 1957. Excursion report Padvindersven 3-7-1957. Staatsbosbeheer. (in Dutch).

Westhoff, V. and den Held, A.J., 1969. Plantengemeenschappen in Nederland. Thieme and Cie, Zutphen, 417 pp. (in Dutch).

Wium-Andersen, S., 1971. Photosynthetic uptake of free CO_2 by the roots of *Lobelia dortmanna*. Physiol. Plant. 25: 245-248.

7

Perspectives for Restoration
A Synthesis

M.J.S. Bellemakers, 2000. Perspectives for restoration: a synthesis.

Justification

In this chapter a general discussion of this thesis will be presented. Before that, I would like to reflect shortly on the use of restoration techniques as a method to reverse the man-induced ecological degradation of shallow surface waters.

First, I would point to the distinction between reversal and restoration of ecosystem developments. In the case of reversal an ecosystem spontaneously regenerates from man-induced ecological degradation after termination of import of pollutants or removal of pollutants. When an ecosystem is affected too much by pollutants to regenerate spontaneously, restoration techniques are required to reverse the effects of the pollutants. In retrospect I think that most of the described ecosystems in this thesis have been degraded so much that spontaneous regeneration after removal of the pollutants does not occur and thus additional restoration techniques are necessary.

However, I think that restoration techniques can not be the sole solution for reinstating the acidified and eutrophicated shallow surface waters in proper state. It is important to remove or suppress the major causes of acidification and eutrophication, *i.e.* the high atmospheric loads of sulphur and nitrogen, before starting restoration. Since 1986 the decrease of the deposition of SO_2 was ca. 70%, the decrease of NH_x was 30-40% and NO_x remained constant. The decrease of total potential acid deposition was ca. 50%. The question remains if this will be sufficient. Otherwise, restoration measures may suppress the symptoms on the short term, but will not provide a long-term solution to acidification and eutrophication of the system.

Yet, even if the sulphur and nitrogen deposition would be suppressed, many of the effects of acidification and eutrophication of the past will continue to exist. This is due to a number of persistent ecosystem changes caused by acidification and eutrophication, for instance, changes in the sapropelium layer. As a result, some rare communities characteristic for shallow surface waters in the Netherlands are severely threatened at present (Arts *et al.*, 1990). The decline of these communities is further enhanced, because the natural causes that create and maintain their habitats (drift-sand habitats) are at present no longer operational, as a consequence of the tendency of earlier management to stabilize existing environments. Restoration techniques which aim at the reversal of these ecosystem changes may be required to prevent the extinction of these communities.

Restoration projects usually have several clear and direct goals, such as the preservation of rare and characteristic species. For this purpose, it may be required to re-establish an ecosystem state of the past. This poses several fundamental questions such as:

- What state do we choose as a reference?
- Can we re-establish the appropriate environmental conditions, and how stable are they?
- What successional changes take place after restoration?

- Should we take continued measures to control succession?

These questions are related to the goal of restoration projects in general. Later on in this chapter, these questions will be answered in relation to other kind of restoration projects.

Methodology and models of restoration ecology

In the introduction of this thesis I have stated some questions about how to obtain a clear idea of the possibilities for restoration of shallow, originally oligotrophic surface waters. This has led to a procedure, which shows a proper methodology of restoration of threatened ecosystems. This model was previously presented by Den Hartog (1993) and later on described by Hobbs and Norton (1996). The model can be defined as a set of key steps in restoration ecology that are considered essential for the successful integration of restoration into land management (quote Hobbs and Norton, 1996):

- Identify processes that have led to degradation or decline.
- Develop methods to reverse or ameliorate the degradation or decline.
- Determine realistic goals for reestablishing species and functional ecosystems, recognizing both the ecological limitations on restoration and the socioeconomic and cultural barriers to its implementation.
- Develop easily observable measures of success.
- Develop practical techniques for implementing these restoration goals at a scale commensurate with the problem.
- Document and communicate these techniques for broader inclusion in land use planning and management strategies.
- Monitor key system variables, assess progress of restoration relative to the agreed-upon goals, and adjust procedures if necessary.

They argue that these steps are essential if ecological restoration is to develop into a useful and widely applicable science. They also mention that recovery of an ecosystem wil not be achieved by removing only one stressor. It, therefore, becomes essential to recognize whether restoration can be achieved by simple removal of stressors and whether this alone will not be sufficient and further actions are needed.

The prevailing paradigm in restoration ecology involves returning a degraded system to some desired state by accelerating biotic change or reinitiating successional processes (Luken, 1990; Edwards *et al.*, 1993). This paradigm leads to the traditional view of restoration options for a degraded system (Figure 1). It illustrates the idea that the ecosystem can travel along a number of different trajectories and that the goal of restoration is to hasten the trajectory towards the desired state. In this view, the past history of the system is not considered, yet the route by which the system reached the present point can have a large impact on the potential for restoration (Hobbs and Mooney, 1993).

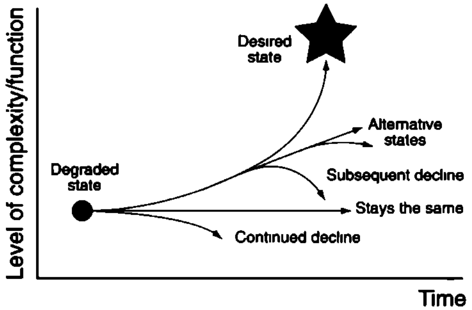


Figure 1: Traditional view of restoration options for a degraded system (redesigned from: Hobbs and Norton, 1996).

Recently, some studies showed conceptual models using a so called "state and transition approach" to examine the restoration of degraded rangeland areas (MacLeod *et al.*, 1993; Ash *et al.*, 1993; Grice and McIntyre, 1995). This model is a hypothetical system that exists in four alternative stable states (Figure 2). State 1 is undegraded, states 2 and 3 are partially degraded and state 4 is highly degraded. Transitions from state 1 to other states occur in response to different stressors or different levels of the same stressor. Transitions back from states 2 and 3 to state 1 are possible if the stressor is removed. But the transition from state 2 (or 3) to state 4 involves crossing a treshold that precludes return to state 2 (or 3) without increased (pro-active) management intervention, even if the stressor is removed.

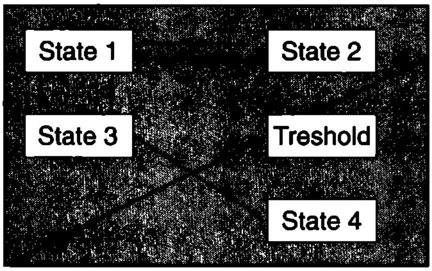


Figure 2: A state and transition approach to restoration (redesigned from: Hobbs and Norton, 1996).

Figure 3 shows the degree of effort required to force transitions between the previous described states. The process of degradation may force transitions that are much more difficult to force back during the process of restoration (Hobbs and Norton, 1996).

The last approach (state and transition) seems comparable with the used methodology and the results of our restoration studies. In the next sections I will try to compare our studies to the state and transition model, show the similarities and explain the differences.

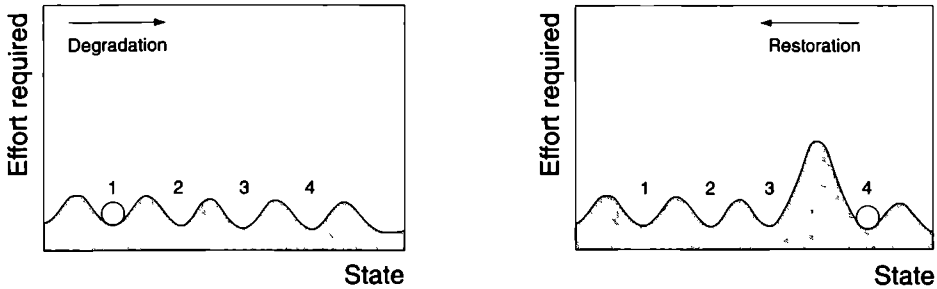


Figure 3: Illustrations of the degree of effort required to force transitions between states (redesigned from: Hobbs and Norton, 1996).

Restoration studies

As a result of the outcome of studies of Schuurkes (1987), Van Dam (1987), Leuven (1988), Arts (1990) and Roelofs (1991[a]) some questions about the restoration possibilities of shallow surface waters have been developed. The major aim of this study, described in the introduction of this thesis, was to investigate the possibilities of restoration of acidified and/or eutrophicated shallow, originally weakly buffered surface waters. The general history of acidification and eutrophication of shallow surface waters in the Netherlands has been summarized in Figure 4. As earlier stated and also concluded from our studies, it is difficult to generalize the used study objects in this way. The combination of acidification and eutrophication by atmospheric deposition (polluted with both sulphur and nitrogen) explains the shown development.

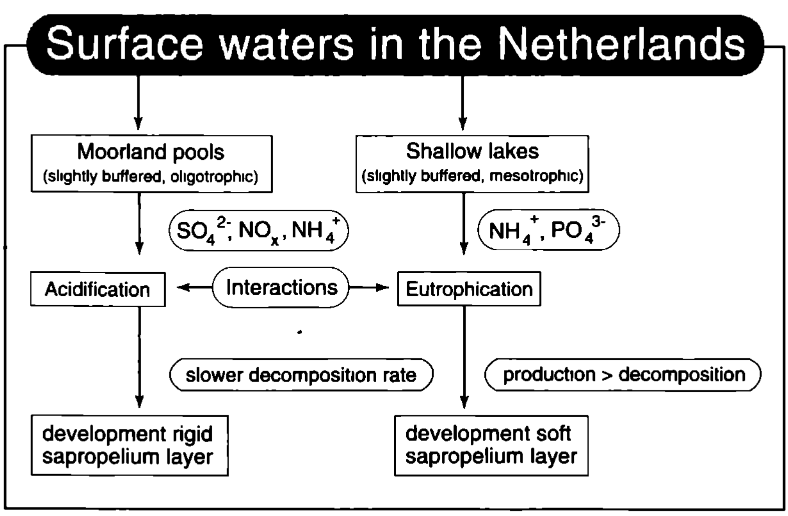


Figure 4: A schematic overview of the history of acidification and eutrophication in shallow surface waters in the Netherlands.

In order to investigate the possibilities of restoration, several studies have been carried out (chapter 2 - 6). To restore acidified or eutrophicated moorland pools, it is necessary to restore the original water chemistry. Restoration can only be successful, if this can be realized. Apart from controlling the atmospheric deposition it is necessary to carry out additional restoration measurements like adding buffering substances or to let in buffered water and remove the organic sapropelium layer.

Enclosure experiments

Enclosure experiments were carried out, before conducting whole lake experiments, to study possible impacts of increased pH and alkalinity (liming) on water chemistry. The effects of liming on the water chemistry have been monitored in two shallow moorland pools (the Ven bij Schaijk and the Padvindersven, chapter 2). The major goal of these experiments was to study the effects of liming on water chemistry in very shallow waters and the eventual processes, triggered by this action, in particular chemical processes, such as internal eutrophication and reacidification.

In acidified moorland pools, an organic sapropelium layer has been formed, due to a decrease of decomposition (Traaen, 1980, Mc Kinley and Vestal, 1982, Rao and Dutka, 1983, Van Dam and Buskens, 1993, Figure 4). In the Ven bij Schaijk liming of the water layer was monitored without removal of the organic sapropelium layer. On the other hand, in the Padvindersven liming of the water layer was monitored after removal of the organic sapropelium layer. The weakly buffered, oligotrophic water conditions returned after liming the water layer in the enclosures in the Ven bij Schaijk and no signs of internal eutrophication were observed. In contrast to these results, liming of the enclosures in the Padvindersven resulted in increased concentrations of ortho-phosphate in the water layer (internal eutrophication), especially when the organic sapropelium layer was not removed. The ortho-phosphate, stored in the organic sapropelium layer of the Padvindersven, was released by the increased pH and alkalinity. Similar decomposition processes were previously described by Kok and Van de Laar (1991).

To investigate internal eutrophication (Mortimer, 1971, Forsberg, 1989, Roelofs, 1991[b]), a similar enclosure experiment was carried out in the Naardermeer, an eutrophicated, formerly mesotrophic shallow lake (chapter 3). In this experiment, the role of inlet water, rich in bicarbonate and sulphate, but poor in ortho-phosphate was tested in two enclosure experiments. This seemed relevant, because of the high concentrations of ortho-phosphate in the water layer of the Naardermeer, despite of inlet of river water, poor in phosphorus.

The decrease of the phosphorus load of the inlet water as a result of phosphate removal did not improve water chemistry in this short-term experiment. Removal of alkalinity from the water layer, however, led to a significant improvement of the water chemistry. The increase of the sulphate flux to the sediment of the Naardermeer caused a decline in the water chemistry probably as a result of the chemical transformations of sulphate in the reductive sapropelium layer. These observations were important since the sulphate concentrations of the inlet water of this lake are very high nowadays. Fortunately, during the years after this experiment until now, water chemistry and water quality of the Naardermeer improved significantly. This improvement can not be deduced from these studies. Recently, other factors such as removal of the sapropelium layer and the removal of large pikes (*Esox lucius* L.) and bream (*Abramis brama* L.) in combination with the inlet of water after phosphate removal have improved the water quality significantly (Meijer, 2000).

Combining the results of the enclosure experiments it is concluded that the release of nutrients was mainly regulated by pH, bicarbonate and sulphate concentrations of the water layer. Kok and Van de Laar (1991) found that an increase of pH and alkalinity in the water layer of acidified moorland pools stimulated the microbial activity in the sediment, whereas according to Carpenter (1980) a decrease of pH and alkalinity in the water layer of eutrophicated, alkaline surface waters inhibited microbial activity in the sediment. The experiments described in this thesis confirm those findings. The increase of nutrients in the water layer after liming without removal of the organic sapropelium layers in acidified moorland pools, such as in the Padvindersven, is more or less similar to the complex situation of the Naardermeer. Therefore, internal eutrophication after increasing the pH and bicarbonate concentrations of the water layer and sediment can be a serious threat after liming of shallow surface waters and needs always to be considered.

These conclusions are comparable within the state transition model of Hobbs and Norton (1996). The treshold of the acidified moorland pools is the organic sapropelium layer, which has to be removed, otherwise a return to a less degraded state is not possible.

Laboratory experiments

To test the above results under more standardised conditions, laboratory experiments were carried out to study the development of macrophytes, characteristic of shallow, oligotrophic, slightly buffered moorland pools (chapter 4). The impacts of both liming and the removal of the organic sapropelium layer were studied in these experiments.

In the culture experiment an increase of the ortho-phosphate concentration in the water layer was observed after adding buffering substances, independent of the removal of the organic sapropelium layer. The increased pH and alkalinity probably caused changes in microbial activity and/or chemical binding of the phosphorus (Brock *et al.*, 1985; Leuven and Wolfs, 1988), resulting in enhanced phosphate concentrations in the water layer (Jackson and Schindler, 1975; Boström *et al.*, 1982; Baccini, 1985). This should be considered as a confirmation of the results of the enclosure experiments.

In both culture experiments the submerged macrophyte *Juncus bulbosus* outcompeted other species characteristic of shallow, oligotrophic, slightly buffered moorland pools (*e.g. Littorella uniflora* and *Lobelia dortmanna*) after liming the water layer. Similar observations were made by Roelofs *et al.* (1994) in lakes in Norway, where after liming a massive expansion of *J. bulbosus* took place. In these studies no enhanced nutrient concentrations in the water layer were found. The nitrogen, phosphorus and carbon dioxide levels in the sediment pore water had increased very strongly and *J. bulbosus* benefited from these circumstances (Wetzel *et al.*, 1985).

A seed bank experiment clearly revealed the good perspectives for restoration of acidified and eutrophicated shallow surface waters. After treatment with clean, artificial rain water, plants developed from the seed bank. Among them were rare isoetid macrophytes, characteristic of shallow, oligotrophic, slightly buffered moorland pools (*e.g. Littorella uniflora* and *Lobelia dortmanna*). These experiments showed that there are still fertile seeds of isoetids present in the seed bank, although those plant species were not found for decades in the sites from which the samples had been taken. This means that the seeds still persist in the seed bank after several decades of acidic or eutrophic conditions and thus, the original vegetation of these ecosystems may be restored, if the right abiotic conditions are reinstated.

Otherwise formulated: if it is possible to remove the treshold, in this case the organic sapropelium layer, many ecosystems can be restored. If the organic sapropelium layer has been removed, less control management is needed to achieve the pre-described goals. In our case, this kind of control management sets back the succession of Littorellion pioneer vegetations. This seed bank experiment is important because it shows whether the system is still viable, or whether it has to be repopulated from sources elsewhere (which may take a long time).

Regeneration after reduction of sulphur and nitrogen in atmospheric deposition

Before discussing the results for restoration of ecosystems, I would like to summarize the effects after reduction of sulphur and nitrogen in atmospheric deposition. Emission reductions and reduced input of industrial, agricultural or civilian pollutants have led to a partial recovery of water quality in some softwater lakes. These are the most desired restoration measures and they show the recovering capacity of softwater lakes. Only ecosystems which have not degraded beyond the treshold (state and transition model) can regenerate to earlier, predefined states, without coarse interventions (spontaneous regeneration, *i.e.* reversibility).

Recovery from acidification

During the past two decades considerable reductions in sulphur emission have been accomplished in large parts of Europe and North America. This has led to a decrease in sulphate and base cation concentrations in many streams and lakes in these areas (Marnette *et al.*, 1993, van Dam and Mertens, 1995, Mattson *et al.*, 1997). Wright *et al.* (1988) report a decrease in sulphate and nitrate runoff within a few weeks after reducing deposition above a catchment. From some soft water lakes, small pH rises, less than 0.5 unit, are reported (Battarbee *et al.*, 1988, Gunn and Keller, 1990, Bouchard, 1997). After reduction of sulphur emissions, the pH of Clearwater lake in the Sudbury- region (Canada) rose from 4.3 to 5.1, leading to a recovery of algal communities (Gunn *et al.*, 1995). pH-related recovery of algal communities has also been observed in Scotland and the Netherlands (Battarbee *et al.*, 1988, van Dam, 1997). However, no recovery of softwater macrophytes has been reported from these lakes. Regeneration is more obvious when the catchment soil and/or the lake sediment is not acidified. Pietsch (1996) describes the recolonization of extremely acid (pH 2-3) pools on base-rich mine spoil within several decades. A pH rise from 4 to 7 has been observed after reducing sulphur input in small artificial softwaters on non-acidified sediment (Brouwer *et al.*, 1996).

Recovery from eutrophication

Many examples of ecosystem recovery after reducing local eutrophication exist (Schindler, 1974). Phosphorus release from the sediment strongly inhibits recovery of the water quality (Marsden, 1989). In most cases, water transparency is enhanced as a consequence of reduced algal growth. This enables the colonization of deeper sediments. Especially characean species have a low light-compensation point and can rapidly colonize such bare, anoxic sediments. Recovery from eutrophication in soft water lakes is generally more successful than in alkaline lakes. This is a result of the low carbon- and phosphorus availability in softwater lakes (Roelofs *et al.*, 1996).

Recovery from nitrogen deposition

The recovery after reduced atmospheric nitrogen deposition has only been studied experimentally. Accumulation of organic matter strongly inhibits recovery of the vegetation. Ten years after ending artificial ammonium sulphate deposition, the former softwater macrophyte vegetation in small artificial ponds on non-acidified soils is still dominated by *Sphagnum*, despite recovery of the water quality (Brouwer *et al.*, 1996). Nitrogen saturation in the catchment soil can persist for many years following the ending of ammonium deposition (Boxman *et al.*, 1998). This is in sharp contrast with the 45% decrease of sulphur storage in the catchment soil, three and a half years after ending sulphate deposition (Wright *et al.*, 1988).

Whole lake experiments

To upscale the results of the previously described laboratory and field experiments, whole lake restoration experiments were done. In this way it was possible to apply the findings of these experiments on ecosystem level and to see if the learned experiences could be extrapolised to practice.

The studied surface waters (chapter 2, 3 and 4) have been classified into four groups according to the following criteria: (i) their original and present water chemistry, (ii) their vegetation and (iii) the cause of their deterioration (Bellemakers *et al.*, 1993). Depending on these criteria, different control measures against the effects of acidification and eutrophication have been selected, executed and biomonitored. Four ecosystems (one of each group) were selected for these ecosystem restoration studies.

The impacts of liming and the removal of the organic sapropelium layer on water chemistry, diatom composition and the fungal infection rate of the moor frog (*Rana arvalis* Nilsson) are described and presented in chapter 5. The effects of restoration measures, including inlet of buffered water, on the development of the vegetation, characteristic of shallow, oligotrophic, slightly buffered moorland pools, in combination with the changes in water chemistry are presented in chapter 6.

Restoration of acidified shallow surface waters

The results of the whole lake experiments confirmed that removal of the organic sapropelium layer of acidified or eutrophicated surface waters is necessary, in order to create good circumstances for restoration. It became obvious that in the first years after removal of the organic sapropelium layer and liming treatments, no indication of re-acidification by accumulated protons as described by Lindmark (1982) in the sediment or by atmospheric deposition took place. Thus, immediately after liming pH rose to pre-acidification levels (the predefined less degraded state) and CO_2 concentrations decreased. This caused a decline of CO_2-using, acid resistant macrophytes and algae (Jackson and Vandermeer, 1990; Alenås *et al.*, 1991). As a consequence of the pH rise, recolonization by acid-sensitive macrophytes and isoetids can occur (Weiher and Boylen, 1994).

The above mentioned side-effects of liming can be avoided when lime (3-20 ton per ha, Traaen *et al.*, 1997) is added to the catchment of a lake. Liming of the catchment area has a slower, but more prolonged effect on the lake water chemistry. A great part of the lime is used to change the base-saturation of the upper soil layers of the catchment to almost 100% (Howells and Dalziel, 1995). Therefore, no annual reacidification can occur. Loss of species in the catchment has not been recorded, except for some *Sphagnum* species. In inflow streams die-off of acidophilous liverworts has been reported (Brettun and Hindar, 1985). Studies at Loch Fleet, Woods lake and Lake Kjallgaard show that a gradual return to pre-liming conditions occurs during the 10 years following liming, thus the liming process has to be repeated (Howells and Dalziel, 1995; Driscoll *et al.*, 1996; Traaen *et al.*, 1997). Only a small decrease in water transparency and no increase of indicators of eutrophicated sediments are reported after catchment liming.

Restoration of eutrophicated shallow surface waters

Mortimer (1971), Forsberg (1989) and Roelofs (1991[a]) described internal eutrophication processes by release of ortho-phosphate from the sediment to the water layer. In our whole ecosystem studies, eutrophication was not observed, on a short term.

Measures against eutrophication in soft water lakes do not differ from the measures taken in other lake types. The reduction of alkalinity in cases where alkalinization is the cause of eutrophication forms an exception. In moderately eutrophicated lakes, slight accumulation of organic matter has occurred on the sediment. When the external nutrient influx to such lakes can be reduced, the eutrophic status and the turbidity of the water layer can decrease (Marsden, 1989; Garnier *et al.*, 1992). This facilitates the recolonization of the largely unchanged sediment. Precipitation of phosphate can be induced temporarily by addition of aluminum or iron salts, but especially when macrophytes are present, a recycling of phosphorus to the water layer occurs in the years following the treatment (Cooke *et al.*, 1993). In cases where a former instream of iron-rich ground water has diminished as a consequence of drainage, the ground water table (in this example the treshold) needs to be restored. This stimulates the precipitation of iron phosphate (iron trap) more permanently. However, in many eutrophicated waters organic matter has accumulated on the sediment. Because of competition with floating macrophytes and algae, most rooting macrophytes have disappeared. The nutrient rich sediment layer is anaerobic and an important source of nutrients and will inhibit recovery of the water quality, especially in shallow lakes (Marsden, 1989; Phillips *et al.*, 1994). Most angiosperms can not easily colonize anaerobic sediments and their presence is often fluctuating due to bioturbation by fish and algal blooms. Restoration measures such as biomanipulation, sediment oxidation and aluminum treatments have been applied in alkaline lakes (Maurizi and Poillon, 1992). These measures can stimulate the colonization of eutrophic sediments by macrophytes, but are insufficient to create a suitable habitat for plant communities of nutrient-poor environments, which are characteristic of the less degraded states. Periodic emergence of the sediment of shallow lakes stimulates germination and colonization by plants. In the Banen (the Netherlands), many soft water macrophytes germinated after reduction of the external nutrient load and after desiccation of the lake floor (Brouwer *et al.*, submitted).

Regeneration of life communities

The whole lake experiments also showed the favourable perspectives on the short term for development of rare macrophytes. Several communities of the plant-alliance Littorellion (*e.g. Littorella uniflora* and *Lobelia dortmanna*) returned or expanded in the treated ecosystems (Table 1). The results of vegetation mapping before and after the restoration measures in the Ven bij Schaijk, Padvindersven and Beuven clearly show the good perspectives on the short term after liming with removal of the organic sapropelium layer (Padvindersven: 4 FLORON red list species returned).

Table 1: The development of some endangered 'FLORON red list species' (Weeda *et al.*, 1990) of three weakly buffered surface waters before and after restoration measures. VS=Ven bij Schaijk; PV=Padvindersven; BV=Beuven; BR=before restoration and AR=after restoration. -=not present; r=rare; +=occasional and ++=local dominant.

Surface water:	VS		PV		BV	
	BR	AR	BR	AR	BR	AR
Ranunculus ololeucos Lloyd	-	-	-	+	-	-
Deschampsia setacea (Hudson) Hackel	-	-	-	-	-	r
Isoetes echinospora Durieu	-	-	-	-	-	r
Apium inundatum (L.) Rchb. f.	-	-	-	-	r	+
Pilularia globulifera L.	-	-	-	-	r	+
Sparganium minimum Wallr.	-	-	-	-	-	+
Lythrum portula (L.) D.A. Webb	-	-	-	+	r	+
Hypericum elodes L.	-	-	-	r	+	++
Littorella uniflora (L.) Aschers.	-	-	-	+	+	++
Lobelia dortmanna L.	-	-	-	-	r	++
Echinodorus repens (Lamk.) Kern & Reichgelt	-	-	-	-	r	++
Luronium natans (L.) Rafin.	-	-	-	-	+	++
Elatine hexandra (Lapierre) DC.	-	-	-	-	+	++

The vegetation development of the Beuven was more spectacular after inlet of buffered water in combination with removal of the organic sapropelium layer (9 FLORON red list species were observed, of which 6 became locally dominant within a few years after the execution of the described restoration measures).

After positive results for water chemistry and vegetation in the first years, liming caused massive *J. bulbosus* expansion within the next years, similar to observations by Roelofs *et al.* (1994) of limed lakes in Norway. Although the Beuven was originally eutrophicated and removal of the sapropelium layer could cause possible acidification, regulated inlet of buffered water did not lead to an overgrowth of *J. bulbosus*. According to Arts *et al.* (1990) further succession of these macrophyte communities could lead to domination of *Drepanocladus fluitans* (Hedw.) Warnst. and *Sphagnum* spec. and further on nymphaeid species (*Nymphaea alba* L. and *Nuphar lutea* (L.) Sm.) as a result of re-acidification. As a result of alkalinization, domination of *Myriophyllum spicatum* L., *Ceratophyllum demersum* L. and *Potamogeton pusillus* L. could occur.

Therefore, inlet of buffered (ground) water seems preferable, because of the better regulation of buffering equivalents and no precipitation of lime on the sediment. This restoration measure should lead to better perspectives for regeneration of acidified or eutrophicated surface waters. On the long term, there is always a risk of internal eutrophication or an overgrowth of *J. bulbosus*. This should be monitored for a long period of time.

In severely eutrophicated lakes, sediment removal can be an effective restoration measure, just as removal of the organic sapropelium layer in acidified shallow waters. For example, sediment removal in the softwater lake Glenwild (United Kingdom) improved oxygenation, increased water transparency and phosphorus availability and phytoplankton biomass decreased (Sebetich and Ferriero, 1997). Removal of accumulated sapropelium strongly stimulates germination of soft water macrophytes (Bellemakers *et al.*, 1996). Nutrient concentrations in the water layer were reduced. After reduction of the external nutrient load and sediment removal in the Beuven and the Banen, a stable community of soft water macrophytes re-established (Brouwer *et al.*, submitted).

Perspectives for restoration

Successful restoration highly depends on proper environmental conditions. Therefore, it is necessary to reduce the polluting agents (sulphur and nitrogen) in the atmospheric deposition to levels beneath the critical loads. Otherwise, it is not possible to achieve on the long term the predefined goals of restoration projects. In terms of the state and transition model (Hobbs and Norton, 1996): if these goals (critical loads) can not be achieved, ecosystems can not be restored into less degraded states because of the pollution.

From the present study, it is concluded that the restoration of shallow, weakly buffered surface waters in the Netherlands may be successful, provided certain conditions are met. These conditions are in practice, apart from reducing the level of pollutants, removal of the sapropelium layer and adding additional buffering substances. The general outline of this study is summarised in Figure 5. It is always necessary to remove the organic sapropelium layer developed under acidic or eutrophic conditions. This organic sapropelium layer is the major treshold in the studied ecosystems, translated to the state and transition model (Hobbs and Norton, 1996). In eutrophicated surface waters, the external load of phosphorus, sulphate and bicarbonate should be reduced. Otherwise, internal eutrophication will occur after having executed the described control measures. Direct liming of the water layer is not an optimal measure, because of the long term effects on water-sediment interactions. Regulated inlet of buffering substances (brook water, ground water) is needed to increase

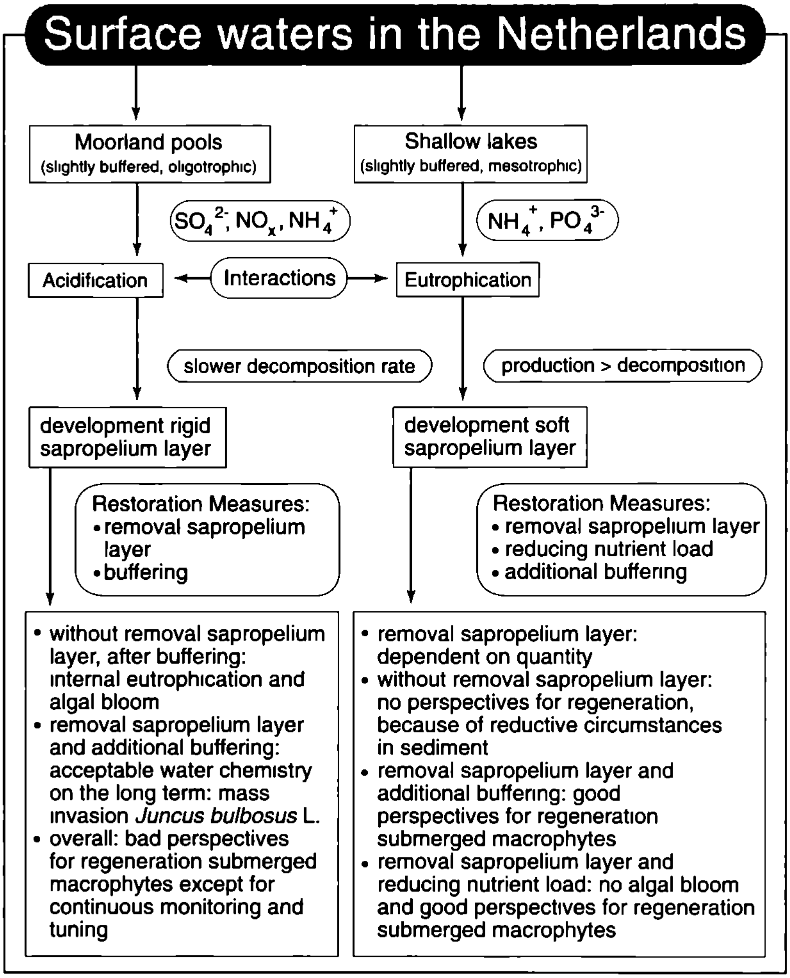


Figure 5: A schematic overview of the results of restoration of shallow surface waters in the Netherlands.

the pH and alkalinity of acidified moorland pools, and to create the proper water chemistry conditions for submerged macrophytes and amphibians.

The seeds persist in the seed banks of the original communities, even for a few decades. Thus, there are good perspectives for regeneration of acidified or eutrophicated surface waters in the Netherlands. In order to realize proper conditions for ecosystem restoration, a valid concept must be followed (Den Hartog, 1993; Hobbs and Norton, 1996). Long term research (additional experiments before and biomonitoring after executing restoration projects) remains necessary, to carry out newly developed approaches to ecosystem manipulation properly and to monitor them.

This approach should be used more in general, instead of the ad hoc, site- and situation-specific approach that now seems to prevail, especially in highly subsidized projects. Control managers should use similar restoration procedures guided by distinct steps, including:

(1) identifying and dealing with the processes leading to degradation in the first place

(2) determining realistic goals and measures of success

(3) developing methods for implementing the goals

(4) incorporating them into management and planning strategies and

(5) monitoring the restoration and assessing its success.

In this way, general guidelines for restoration on individual sites could be based on the concepts of designed disturbance, controlled colonization and controlled species performance. Development of these guidelines is an important priority so that urgent whole ecosystem restoration can be planned and implemented effectively.

Moreover, it should be considered how long the desired effects of the control measures can last in the studied ecosystems. As long as the pollution by atmospheric deposition is not reversed to acceptable levels, the results of the measures have to be monitored intensively and if necessary the control measures have to be repeated.

This study only describes the possibilities of reversibility of the effects of acidification and eutrophication of shallow surface waters in the Netherlands. Although, considering the previous alineas, it should be possible to extrapolate the findings of this study to different types of ecosystems, with all possible variations on restoration techniques.

Acknowledgements

The author is indebted to Prof. Dr. C. den Hartog, Dr. R. Bobbink, Drs. M. Maessen, Dr. J.G.M. Roelofs, Drs. E. Brouwer, Drs. P.J.A. Vervuren and Dr. H. van Dam for the constructive discussions and for critically reading this synthesis.

References

Alenås, I , Andersson, B I , Hultberg, H and Rosemarin, A , 1991 Liming and reacidification reactions of a forest lake ecosystem, lake Lysevatten, in SW Sweden Water, Air, Soil Pollut 59 55-77

Arts, G H P, 1990 Deterioration of atlantic soft-water systems and their flora, a historical account Thesis University of Nijmegen, 197 pp

Arts, G H P, Van der Velde, G , Roelofs, J G M and Van Swaay, C A M , 1990 Successional changes in the soft-water macrophyte vegetation of (sub)atlantic, sandy, lowland regions during this century Freshwat Biol 24 287-294

Ash, A J , McIvor, J G and Brown, J R , 1993 Land condition and overgrazing management paradox for the savannas of northern Australia In Proc of the XVII Int Grassland Congress, 1930-1931 Palmerston North, New Zealand

Baccini, P, 1985 Phosphate interactions at the sediment-water interface In W Stumm (Editor), Chemical Processes in Lakes Wiley, New York, p 189-224

Battarbee, R W , Flower, R J , Stevenson, A C , Jones, V C , Harriman, R and Appleby, P G , 1988 Diatom and chemical evidence for reversibility of acidification of Scottish lochs Nature 332 530-532

Bellemakers, M J S , Maessen, M , Cals, M J R and Roelofs, J G M , 1993 Effectgerichte maatregelen tegen verzuring en eutrofiering van oppervlaktewateren Report monitoringsprogramma eerste fase Department Ecology, Section Environmental Biology, University of of Nijmegen, by order of the Ministry of Agriculture, Nature Conservation and Fisheries, 148 pp (in Dutch)

Bellemakers, M J S , Maessen, M , Verheggen, G M and Roelofs, J G M , 1996 Effects of liming on water chemistry in shallow acidified moorland pools the germination and development of aquatic macrophytes Aquat Bot 54 37-50

Bostrom, B , Jansson, M and Forsberg, C , 1982 Phosphorus release from lake sediments Arch Hydrobiol Beih /Ergebn Limnol 18 5-60

Bouchard, A , 1997 Recent lake acidification and recovery trends in Southern Quebec, Canada Water, Air, Soil Pollut 94 225-245

Boxman, A W , Blanck, K , Brandrud, T E , Emmett, B A , Gundersen, P , Hogervorst, R F , Kjoenaas, O J , Persson, H and Timmermann, V , 1998 Vegetation and soil biota response to experimentally-changed nitrogen inputs in coniferous forest ecosystems of the NITREX For Ecol Man 101 65-80

Brettum, P and Hindar, A , 1985 Effects of lime treatment on the biological system In Baalsrud, K , 1985 Liming of acid water Norwegian department of the Environment, 147 pp

Brock, Th C M , Boon, J J and Paffen, B G P, 1985 The effects of the season and water chemistry on the decomposition of *Nymphaea alba* L , weight loss and pyrolysis mass spectrometry of the particulate matter Aquat Bot 22 197-229

Brouwer, E , Bobbink, R , Meeuwsen, F and Roelofs, J G M , 1996 Recovery from acidification in aquatic mesocosms after reducing ammonium and sulphate deposition Aquat Bot 56 119-130

Brouwer, E , Soontiens, J , Bobbink, R and Roelofs, J G M , submitted Sulphate and bicarbonate as key factors in sediment degradation and restoration of Lake Banen Aquat Conserv

Carpenter, S R , 1980 Enrichment of Lake Wingra, Winsconsin, by submerged macrophyte decay Ecology 61 1145-1155

Cooke, G D , Welch, E B , Peterson F, Martın, A B , Fulmer, D G , Hyde, J B and Schneve, G D , 1993 Effectiveness of Al, Ca and Fe salts for control of internal phosphorus loading in shallow and deep lakes Hydrobiologia 253 323-335

Den Hartog, C , 1993 Effectgerichte maatregelen tegen verzuring en eutrofiering in natuurterreinen In Cals, M J R , De Graaf, M C C and Roelofs, J G M (Eds), Effectgerichte maatregelen tegen verzuring en eutrofiering in natuurterreinen, University of Nijmegen, p 1-5 (in Dutch)

Driscoll, C T , Cirmo, C P , Fahey, T J , Blette, V L , Bukaveckas, P A , Burns, D A , Gubala, C P, Leopold, D J , Newton, R M , Raynal, D J , Schofield, C L , Yavitt, C L and Porcella, D B , 1996 The experimental watershed liming study comparison of lake and watershed neutralization strategies Biogeochem 32 143-147

Edwards, C A , Grove, T L , Harwood, R R and Pierce Colfer, C J , 1993 The role of agroecology and integrated farming systems in agricultural sustainability Agric , Ecosyst and Env 46 99-121

Forsberg, C , 1989 Importance of sediments in understanding nutrient cycling in lakes Hydrobiologia 176/177 263-277

Garnier, J , Chestérikoff, A , Testard, P and Garban, B , 1992 Oligotrophication after a nutrient reduction in a shallow sand-pit lake (Créteil Lake, Paris suburbs, France) a case of rapid restoration Annls Limnol 28 253-262

Grice, A C and McIntyre, S , 1995 Speargrass (*Heteropogon contortus*) in Australia dynamics of species and community Rangelands Journal 17 3-25

Gunn, J M and Keller, W , 1990 Biological recovery of acid lake after reduction in industrial emissions of sulphur Nature 345 431-433

Gunn, J , Keller, W , Negusanti, R , Potvin, R , Beckett, P and Winterhalder, K , 1995 Ecosystem recovery after emission reductions Sudbury, Canada Water, Air, Soil Pollut 85 1783-1788

Hobbs, R J and Mooney, H A , 1993 Restoration ecology and invasions In D A Saunders, R J Hobbs and P R Ehrlich (Eds) Nature conservation 3 reconstruction of fragmented ecosystems, global and regional perspectives 127-133 Surrey Beatty and Sons, Chipping Norton, New South Wales, Australia

Hobbs, R J and Norton, D A , 1996 Towards a conceptual framework for restoration ecology Restor Ecol 4 93-110

Howells, G and Dalziel, T , 1995 A decade of studies at Loch Fleet, Galloway (Scotland) a catchment liming project and restoration of a browntrout fishery Freshwater Forum 15 4-37

Jackson, T A and Schindler, D W , 1975 The biogeochemistry of phosphorus in an experimental lake environment evidence for the formation of humic-metal-phosphate complexes Verh Internat Verein Limnol 19 211-221

Jackson, M B and Vandermeer, E M , 1990 Effects of neutralization and early reacidification on filamentous algae and macrophytes in Bowland lake Can J Fish Aquat Sci 47 432-439

Kok, C J and Van de Laar, B J , 1991 Influence of pH and buffering capacity on the decomposition of *Nymphaea alba* L detritus in laboratory experiments a possible explanation for the inhibition of decomposition at low alkalinity Verh Internat Verein Limnol 24 2689-2692

Leuven, R S E W , 1988 Impact of acidification on aquatic ecosystems in The Netherlands with emphasis on structural and functional changes Thesis University of Nijmegen, 181 pp

Leuven, R.S.E.W. and Wolfs, W.J., 1988. Effects of water acidification on the decomposition of *Juncus bulbosus* L. Aquat. Bot. 31: 57-81.

Lindmark, G., 1982. Acidified lakes: sediment treatment with sodium carbonate - a remedy? Hydrobiologia 92: 537-547.

Luken, J.O., 1990. Directing ecological succession. Chapman and Hall, New York, 437 pp.

MacLeod, N.D., Brown, J.L. and Noble, J.C., 1993. Ecological and economic considerations for the management of shrub encroachment in Australian rangelands. In: Proc. of the 10th Australian Weeds Conf. and 14th Asian Pacific Weed Society Conf. 6-10 September 1993, vol 2: 118-121. Weed Society of Queensland, Brisbane, Australia.

Marnette, E.C.L., Houweling, H., Van Dam, H. and Erisman, J.W., 1993. Effects of decreased atmospheric deposition on the sulfur budgets ot two Dutch moorland pools. Biogeochem. 23: 119-144.

Marsden, M.W., 1989. Lake restoration by reducing external phosphorus loading: the influence of sediment phosphorus release. Freshwat. Biol. 21: 139-162.

Mattson, M.D., Godfrey, P.J., Walk, M.F., Kerr, P.A. and Zajicek, T., 1997. Evidence of recovery from acidification in Massachusetts streams.Water, Air, Soil Pollut. 96: 211-232.

Maurizi, S. and Poillon, F., 1992. Restoration of aquatic ecosystems: science, technology and public policy. National Acad. Press, Washington. 552 pp.

McKinley, V.L. and Vestal, J.R., 1982. Effects of acid on plant litter decomposition in an arctic lake. Appl. Environ. Microbiol. 43: 1188-1195.

Meijer, M.L., 2000. Biomanipulation in the Netherlands: 15 years of experience. Thesis Agriculture University Wageningen, 208 pp.

Mortimer, C.H., 1971. Chemical exchanges between sediments and water in the Great Lakes, speculations on probable regulatory mechanisms. Limnol. Oceanogr. 16: 387-404.

Phillips, G., Jackson, R., Bennet, C. and Chilvers, A., 1994. The importance of sediment phosphorus release in the restoration of very shallow lakes (The Norfolk Broads, England) and implications for biomanipulation. Hydrobiologia 275/276: 445-456.

Pietsch, W.H.O., 1996. Recolonization and development of vegetation on mine spoils following brown coal mining in Lusatia. Water, Air, Soil Pollut. 91: 1-15.

Rao, S.S. and Dutka, B.J., 1983. Influence of acid precipitation on bacterial populations in lakes. Hydrobiologia 98: 153-157.

Roelofs, J.G.M., 1991[a]. Vegetation under chemical stress: effects of acidification, eutrophication and alkalinisation. Thesis University of Nijmegen, 167 pp.

Roelofs, J.G.M., 1991[b]. Inlet of alkaline river water into peaty lowlands: effects on water quality and *Stratiotes aloides* L. stands. Aquat. Bot. 39: 267-293.

Roelofs, J.G.M., Brandrud, T.E. and Smolders, A., 1994. Mass invasion of *Juncus bulbosus* after liming of acidified Norwegian lakes. Aquat. Bot. 48: 187-202.

Roelofs, J.G.M., Bobbink, R., Brouwer, E. and de Graaf, M.C.C., 1996. Restoration ecology of aquatic and terrestrial vegetation on non-calcareous sandy soils in The Netherlands. Acta Bot. Neerl. 45: 517-541.

Schindler, D.W., 1974. Eutrophication and recovery in experimental lakes: implications for lake management.

Science 184 897-899

Schuurkes, J A A R , 1987 Acidification of surface waters by atmospheric deposition Thesis University of Nijmegen, 160 pp

Sebetich, M J and Ferriero, N , 1997 Lake restoration by sediment dredging Verh Internat Verein Limnol 26 776-781

Traaen, T S , 1980 Effects of acidity on decomposition of organic matter in aquatic environments In D Drabløs and A Tollan (Eds) Ecological Impact of Acid Precipitation Proc Int Symp , March 1980, Sandefjord, Norway 340-341 New York, U S A , 780 pp

Traaen, T S , Frogner, T , Hindar, A , Kleiven, E , Lande, A and Wright, R F , 1997 Whole-catchment liming at Tjønnstrond, Norway an 11-year record Water, Air, Soil Pollut 94 163-180

Van Dam, H , 1987 Verzuring, een tijdsverschijnsel Thesis Agriculture University Wageningen, 160 pp

Van Dam, H and Buskens, R F M , 1993 Ecology and management of moorland pools balancing acidification and eutrophication Hydrobiologia 265 225-263

Van Dam, H and Mertens, A , 1995 Long-term changes of diatoms and chemistry in headwater streams polluted by atmospheric deposition of sulphur and nitrogen compounds Freshwat Biol 34 579-600

Van Dam, H , 1997 Vennen herstellen zich gedeeltelijk van verzuring H_2O (30) 11 366- 370

Weeda, E J , Van der Meijden, R and Bakker, P A , 1990 Rode lijst van de in Nederland verdwenen en bedreigde planten (Pteridophyta en Spermatophyta) over de periode 1 1 1980 - 1 1 1990 Gorteria 16 2-26 (in Dutch)

Weiher, E R and Boylen, C W , 1994 Alterations in aquatic plant structure following liming of an acidic Adirondack lake Can J Fish Aquat Sci 51 20-24

Wetzel, R G , Brammer, E S , Lindstrom, K and Forsberg, C , 1985 Photosynthesis of submerged macrophytes in acidified lakes II Carbon limitation and utilization of benthic CO_2 sources Aquat Bot 22 107-120

Wright, R F , Lotse, E and Semb, A , 1988 Reversibility of acidification shown by whole catchment experiments Nature 334 670-675

RESTAURATIE VAN VERZUURDE EN GEËUTROFIEERDE ONDIEPE OPPERVLAKTEWATEREN MOGELIJKHEDEN TOT HERSTEL

Vennen, heiden en stuifzanden zijn typisch Nederlandse, natuurlijke landschapselementen. Ze komen voor op de van oorsprong voedselarme zandgronden, afgezet na de laatste ijstijd. Op deze zandgronden konden zwakgebufferde, voedselarme vennen op natuurlijke wijze ontstaan: door uitstuiving (wind), uitslijping (rivierwater, terugtrekkend ijs) en stagnatie van regenwater. Sommige vennen zijn door menselijke bedrijvigheid ontstaan, bijvoorbeeld voor het wassen van schapen of het maken van zwemvijvers. Door het stoppen van de natuurlijke processen (bebossing, uitstuiving en sedimentatie naar één kant in vennen) ontstaan er niet langer nieuwe vennen.
Vennen worden geheel of gedeeltelijk door regenwater gevoed; daardoor varieert de waterstand. Door de geringe toevoer van voedingsstoffen is de plantengroei er schaars en leven er weinig organismen in het water. Sommige planten vertonen speciale aanpassingen aan het zwakgebufferde en voedselarme (oligotrofe) milieu. Ze behoren tot een bepaalde klasse van gemeenschappen: de Oeverkruidklasse.
De verzuring enerzijds en de verrijking met voedingsstoffen (eutrofiëring) anderzijds van deze ondiepe, zwakgebufferde, voedselarme oppervlaktewateren hebben de afgelopen decennia tot een verslechtering van de waterkwaliteit geleid. De verzuring had een afname van de pH (een maat voor de zuurgraad van het water) en de bicarbonaatconcentratie (buffercapaciteit: vermogen om verzuring te weerstaan) tot gevolg. Deze veranderingen leidden tot de afname van de afbraaksnelheid van afgestorven plant- en diermateriaal, dat op de bodem terecht komt. Hierdoor ontstond na verloop van tijd een dikke sliblaag in verzuurde vennen, bestaande uit afgestorven algen, planten en waterdieren (organische sliblaag). Het teruglopen van de waterkwaliteit was duidelijk zichtbaar aan een afname van het aantal plant- en diersoorten, die de verzuurde vennen (voornamelijk in het zuidoosten en oosten van Nederland) en geëutrofieerde ondiepe oppervlaktewateren (vennen én laagveenplassen in het westen van Nederland) oorspronkelijk bevolkten. Een voorbeeld hiervan is de ernstige achteruitgang van plantensoorten van de Oeverkruidklasse, een karakteristieke groep van plantengemeenschappen van ondiepe, zwakgebufferde, voedselarme oppervlaktewateren.
Het doel van dit onderzoek is om de herstelmogelijkheden, voor zowel de waterchemie als de vegetatie-ontwikkeling van deze ecosystemen te toetsen. Daartoe zijn in de periode 1987 tot en met 1992 verschillende soorten experimenten uitgevoerd, op verschillende schaalniveau's (cylinder-experimenten in het veld, laboratoriumexperimenten en totale ecosysteemexperimenten in het veld).

Als eerste werden in het veld cylinder ('enclosure') experimenten uitgevoerd. 'Enclosures' zijn grote kunststof cylinders, die in de bodem van een ven kunnen worden geplaatst en boven het wateroppervlak uitsteken. Na plaatsing is uitwisseling van water tussen de cylinder en het open water niet langer mogelijk en onstaan er dus mini-vennetjes. Hierin kan de waterlaag gemanipuleerd worden, bijvoorbeeld door het bekalken of door het inlaten van gebufferd water. Daarop volgend kan de waterchemie gevolgd worden, gedurende een langere tijd (monitoring).
Met behulp van deze 'mini-vennen' werden de effecten van bekalking op twee verzuurde vennen bestudeerd (hoofdstuk 2), het ven bij Schaijk en het Padvindersven (bij Rijsbergen). Hieruit kon worden afgeleid dat het toevoegen van kalk tot een toename van de pH en de buffercapaciteit leidde. In het Padvindersven vond na de bekalking een toename van voedingsstoffen (vooral van fosfaat) in het water plaats. Dit gebeurde als de organische sliblaag niet verwijderd werd. Dit proces, waarbij geen voedingsstoffen van buitenaf aangevoerd worden, wordt interne eutrofiëring genoemd. Na de verhoging van de afbraaksnelheid door de verhoogde pH en buffercapaciteit, kwamen deze voedingsstoffen vanuit de organische sliblaag in de waterlaag vrij.
Deze interne eutrofiëring trad ook op in het oudste natuurreservaat van Nederland: het Naardermeer. Om watertekorten, als gevolg van de toenemende verdroging, aan te vullen werd al vanaf 1975 besloten rivierwater in te laten. Dit inlaatwater kwam uit een nabij gelegen rivier: de Vecht. Hierdoor nam de hoeveelheid voedingsstoffen in de waterlaag van het Naardermeer toe: de waterkwaliteit werd bedreigd door aanvoer van fosfaten van buitenaf (externe eutrofiëring). Om die reden werden sinds 1985, alvorens het water in te laten, de aanwezige fosfaten tot een aanvaardbaar niveau teruggebracht (defosfatering). Omdat er al lang water vanuit de Vecht ingelaten werd, waren er grote hoeveelheden nutriënten in het organische slib opgeslagen. Dit werd enerzijds veroorzaakt door wegzijging (water dat via de enigszins doorlaatbare bodem naar beneden wegzakt) en anderzijds door de verhoogde groei van algen die de afbraak van algen ruimschoots overtrof. Door de mobilisatie van fosfaten van het organische slib naar de waterlaag was het water van het Naardermeer na inlaat van fosfaat-arm water nog steeds voedselrijk (interne eutrofiëring). Om deze eutrofiëring beter te kunnen begrijpen, werden ook in het Naardermeer een tweetal cylinder experimenten uitgevoerd (hoofdstuk 3).
Uit historisch onderzoek en deze cylinderexperimenten kwam naar voren dat naast de verhoogde fosfaatconcentratie, ook de pH, bicarbonaat- , en vooral sulfaatconcentraties sterk waren toegenomen in het Naardermeer. Dit was een gevolg van de inlaat van het bicarbonaat- en sulfaatrijke rivierwater. De verhoogde pH en bicarbonaatconcentratie leidden tot een toename van de afbraaksnelheid van het organisch materiaal en dus tot interne eutrofiëring. De toename van de sulfaatconcentratie kan ook een rol spelen in de interne eutrofiëring. Het sulfaat kan in een sliblaag omgezet worden in

sulfide (sulfaatreductie), een giftige stof Hierdoor hebben in de sliblaag wortelende waterplanten geen mogelijkheden om zich te kunnen ontwikkelen Sulfaatreductie is een zuurstof verbruikend proces en het hierbij gevormde sulfide slaat neer met ijzer Hierdoor komt het fosfaat, normaliter grotendeels aan het ijzer gebonden, vrij en kan zich daarna gemakkelijker van de sliblaag naar de waterlaag verplaatsen

Door de resultaten van de cylinderexperimenten in de verzuurde vennen enerzijds en die van het Naardermeer anderzijds te combineren, kan geconcludeerd worden dat de pH en de buffercapaciteit belangrijke, sturende factoren van interne eutrofieringsprocessen zijn Verhoging van de pH en de buffercapaciteit in verzuurde vennen en in laagveenplassen veroorzaakt een verhoogde beschikbaarheid van fosfaät in de waterlaag Interne eutrofiering kan derhalve een belemmering vormen voor een gunstige ontwikkeling van de waterkwaliteit na de uitvoering van beheersmaatregelen in oppervlaktewateren, zoals het bekalken van verzuurde vennen of het inlaten van bicarbonaat-, fosfaat-, en sulfaatrijk water in laagveenplassen

Om de resultaten van de cylinderexperimenten in de verzuurde vennen beter te kunnen begrijpen, zijn er experimenten uitgevoerd onder meer geconditioneerde omstandigheden in het laboratorium en in een kas Gedurende vier maanden is het verloop van de biomassa (een maat voor de organische groei) van twee waterplanten bij verschillende buffercapaciteiten onderzocht (hoofdstuk 4) Hierbij is gekozen voor twee soorten, die van oudsher vennen karakteriseren het bedreigde oeverkruid (*Littorella uniflora* (L) Aschers) en knolrus (*Juncus bulbosus* L), een soort die bij verzuring andere soorten uit het Oeverkruidverbond overwoekerde Uit deze proef bleek dat ook na bekalking knolrus enorm kan gaan woekeren Dit soort woekering was al eerder waargenomen na bekalking van Noorse meren

Na bekalking van de waterlaag boven enkele, uit verschillende vennen gehaalde sedimenten, overwoekerde knolrus eveneens de overige plantensoorten, vooral als de organische sliblaag niet werd verwijderd Na het verwijderen van deze sliblaag werden in veel gevallen zeldzame, in de betreffende vennen reeds lange tijd niet meer geobserveerde plantesoorten teruggevonden Voorbeelden zijn het oeverkruid van het Padvindersven (sinds 1957 niet meer waargenomen) en de waterlobelia (*Lobelia dortmanna* L) van het Peetersven bij Eindhoven (sinds 1966 niet meer waargenomen) De zaden van deze specifieke soorten van de Oeverkruidklasse waren dus nog vitaal in de zaadbank aanwezig

Deze waarnemingen leidden tot de conclusie dat er mogelijkheden bestaan om verzuurde vennen te herstellen na het kunstmatig verhogen van de pH en de buffercapaciteit De waterchemie kan pas optimaal hersteld worden na verwijdering van de organische sliblaag Daardoor worden er gunstige omstandigheden voor kieming, vestiging en handhaving van plantensoorten, karakteristiek voor dit

milieu, gecreëerd. Met de sliblaag wordt naast de voorraad knolruszaden ook een deel van de overtollige hoeveelheid voedingsstoffen verwijderd. Daarbij komt een oudere zaadvoorraad aan de oppervlakte, met zaden van planten die oorspronkelijk in deze ecosystemen voorkwamen. Deze zaden kunnen, mits de juiste voorwaarden gecreëerd worden, ontkiemen, ondanks het feit dat zij in sommige gevallen al tientallen jaren geleden in die zaadvoorraad terecht gekomen zijn.
Om deze resultaten terug te koppelen naar het veld, zijn er voor 15 verschillende locaties beheersmaatregelen voorgesteld en uitgevoerd. Van vier oppervlaktewateren (de Tongerense Heide bij Zwolle, het ven bij Schaijk, het Padvindersven bij Rijsbergen en het Beuven bij Eindhoven) zijn de beheersmaatregelen met betrekking tot de effecten van verzuring en eutrofiëring in dit proefschrift beschreven (hoofdstuk 5 en 6). In deze beschrijvingen zijn de resultaten van een uitvoerige biomonitoring (het volgen van de waterchemie- en de vegetatieontwikkeling vóór en ná de uitvoering van de beheersmaatregelen) weergegeven.
Na verwijdering van de organische sliblaag, om interne eutrofiëring en/of woekering van knolrus te voorkomen waren de resultaten hoopvol, vooral op korte termijn . In het Padvindersven werden binnen één jaar na verwijdering van de sliblaag en de daarop volgende bekalking maar liefst vier zeer ernstig bedreigde plantesoorten (FLORON rode lijst van bedreigde plantesoorten) aangetroffen. Echter, op een wat langere termijn bleek ook hier knolrus zich massaal te ontwikkelen, net als in bekalkte meren in Noorwegen. Daar leek de waterchemie en plantengroei zich in eerste instantie positief te ontwikkelen, maar na enkele jaren werden meters dikke pakketten knolrus in de waterlaag waargenomen.
In de wateren waar de sliblaag voor bekalking niet verwijderd werd trad een verhoging op van het voedingsstoffengehalte, als gevolg van eutrofiëring. Dit was af te leiden uit wijzigingen in de soortensamenstelling van kiezelwieren (ééncellige waterplanten). Daarnaast werd er in de kleine heidevennetjes van de Tongerense Heide een positieve bijwerking gevonden van de bekalking. Door de verzuring was het beschimmelingspercentage van de eieren van de heikikker (*Rana arvalis* Nilsson) significant toegenomen. Na bekalking liep het beschimmelingspercentage sterk terug (hoofdstuk 5).
Om de negatieve lange termijn ontwikkelingen tegen te gaan, werd in het Beuven na verwijdering van de organische sliblaag een andere wijze van bufferen toegepast. In het ven werd voorgezuiverd beekwater (weinig voedingsstoffen bevattend) met een relatief hoge buffercapaciteit ingelaten. Op deze wijze is het mogelijk om de hoeveelheid in te laten bufferstof nauwgezet te reguleren. De resultaten waren verbluffend: alle oorspronkelijke plantesoorten van de oeverkruidklasse keerden terug in hoge bedekkingspercentages. Negen daarvan kwamen op de rode lijst van extreem bedreigde plantesoorten voor. Door de uitvoering van deze beheersopties lijkt het Beuven weer wat meer op

het Beuven zoals het decennia voor de herstelmaatregelen er uitgezien moet hebben en waar het van oudsher haar gròte bekendheid aan te danken had.

Naar aanleiding van het totale onderzoek kan geconcludeerd worden dat er goede herstelmogelijkheden zijn voor ondiepe, zwakgebufferde, voedselarme wateren. Echter, de belangrijkste herstelmaatregel verdient extra aandacht: het terugdringen van de luchtverontreiniging, de veroorzaker van zowel verzuring als eutrofiëring. Mocht daar aan voldaan worden, dan nog is het noodzakelijk aanvullende maatregelen in verzuurde vennen uit te voeren, zoals verwijdering van de sliblagen en het eventueel licht bufferen van de waterlaag. In laagveenplassen zal de externe aanvoer (inlaat) van bicarbonaat- en voedingsstoffenrijk (fosfaat en sulfaat) water drastisch verminderd moeten worden. In beide systemen zal anders interne eutrofiëring kunnen optreden. Daarnaast kan zich in de verzuurde vennen een explosieve ontwikkeling van knolrus voordoen. De buffering zal gereguleerd plaats moeten vinden, zoals mogelijk is door inlaat van voorgezuiverd beekwater of bufferend grondwater. Alleen dan kunnen de gunstige randvoorwaarden gecreëerd worden voor de karakteristieke levensgemeenschappen, thuishorend in dit type oppervlaktewateren.

LIST OF PUBLICATIONS

Publications in peer-reviewed, international journals

M.J.S. Bellemakers and H. van Dam (1992). Improvement of breeding success of the moor frog (*Rana arvalis*) by liming of acid moorland pools and the consequences of liming for water chemistry and diatoms. Environ. Pollut. 78: 165-171.

M.J.S. Bellemakers, M. Maessen and J.G.M. Roelofs (1994). Effects of liming on water chemistry in shallow acidified pools in the Netherlands: enclosure experiments. Water, Air, Soil Pollut. 73: 131-142.

M.J.S. Bellemakers (1994). Restoration of the Littorellion vegetation of shallow, slightly buffered, oligotrophic moorland pools. Acta Bot. Neerl. 43: 393-394.

M.J.S. Bellemakers, M. Maessen, G.M. Verheggen and J.G.M. Roelofs (1996). Effects of liming on shallow acidified moorland pools: a culture and seed bank experiment. Aquat. Bot. 54: 37-50.

M.J.S. Bellemakers and M. Maessen (1998). Effects of alkalinity, acidity and external sulphate and phosphorus load on water chemistry in the 'Naardermeer'. Water, Air, Soil Pollut. 101: 3-14.

M.J.S. Bellemakers, M. Maessen, R. Bobbink and J.G.M. Roelofs (submitted). Restoration of acidified and eutrophicated shallow surface waters. Restor. Ecol.

A.W. Boxman, H. Krabbendam, M.J.S. Bellemakers and J.G.M. Roelofs (1991). Effects of aluminium on the development and nutrition of *Pinus nigra* in hydroculture. Environ. Pollut. 73: 119-136.

M. Maessen, J.G.M. Roelofs, M.J.S. Bellemakers en G.J.M. Verheggen (1992). The effects of aluminium, aluminium/calcium ratios and pH on aquatic plants from poorly buffered environments. Aquat Bot. 43: 115-127.

H. van Dam and M.J.S. Bellemakers (1992). Diatoms as indicators for the management of moorland pools. Acta Bot. Neerl. 40: 378-379.

Other publications

M.J.S. Bellemakers, M. Maessen and G.M. Verheggen (1990). Restauratie van verzuurde en geëutrofieerde zwakgebufferde ondiepe oppervlaktewateren; mogelijkheden tot herstel. eindrapport i.o.v. het Ministerie van Volkshuisvesting, Ruimtelijke Ordening & Natuurbeheer, 96 pp.

M J S Bellemakers and H van Dam (1990) Impact of liming on chemistry, diatoms, macrophytes and moor frogs (*Rana arvalis*) in acid shallow moorland pools in the Netherlands Conference Abstracts of the International Conference on Acidic Deposition Its Nature and Impacts 251 pp

M J S Bellemakers and M Maessen (1991) Mergel en bulldozer bieden soelaas voor verzuurde vennen Goede hoop op verbetering waterkwaliteit, flora en fauna Tijdschrift voor ruimtelijke ordening en milieubeheer 8 13-16

M J S Bellemakers, H van Dam and A J M Roozen (1991) Kan de Heikikker worden behouden door bekalking van vennen? De Levende Natuur 6 228-232

M J S Bellemakers (1991) Ecofysiologie van waterplanten In Stichting River Research (red), Syllabus Cursus Ecologie van water- en moerasplanten, p 16-21

M J S Bellemakers and M Maessen (1991) Onderzoek naar de herstelmogelijkheden van verzuurde en geeutrofieerde zachte wateren In Stichting River Research (red), Syllabus Cursus Ecologie van water- en moerasplanten, p 48-56

M J S Bellemakers, M Maessen, M J R Cals and J G M Roelofs (1993) Effectgerichte Maatregelen tegen verzuring en eutrofiering van oppervlaktewateren Eindrapport monitoringsprogramma eerste fase Eindrapport i o v het Ministerie van Landbouw, Natuurbeheer & Visserij, pp 148

M J S Bellemakers and D J de Jong (1995) De verspreidingsdynamiek van zeegras in de Oosterschelde de effecten van het Deltaplan Notitie in opdracht van RIKZ-RWS Middelburg en NIOO-CEMO Yerseke, 50 pp

M J S Bellemakers, G Hoogerwerf, R Krekels and P J M Verbeek (1995) Monitoring- en evaluatieplannen voor vier natuurvriendelijke oeverprojecten (MEP-4) Adviesbureau Natuurbalans/Limes Divergens in opdracht van Directie Oost-Nederland, Rijkswaterstaat 50 pp

M J R Cals, J G M Roelofs, M J S Bellemakers, M Maessen, M C C de Graaf and P J M Verbeek (1990) Monitoring van effectgerichte maatregelen tegen verzuring en eutrofiering in oppervlaktewateren en heide-schraallanden Interim rapport 1990, 45 pp

M J R Cals, J G M Roelofs, M J S Bellemakers, M Maessen, M C C de Graaf and P J M Verbeek (1991) Monitoring van effectgerichte maatregelen tegen verzuring en eutrofiering in oppervlaktewateren en heide-schraallanden Interim rapport 1991, 62 pp

M J R Cals, M J S Bellemakers, M Maessen and J G M Roelofs (1993) Voorwaarden en perspectieven voor herstel van verzuurde en geeutrofieerde oppervlaktewateren In Effectgerichte maatregelen tegen verzuring en eutrofiering in natuurterreinen, M Cals, M de Graaf en J Roelofs (Red), Universiteit van Nijmegen p 31-62

M Maessen and M J S Bellemakers (1992) Restoration of shallow acidified lakes in the Netherlands Proceedings Experimental manipulations of biota and biogeochemical cycling in ecosystems May 18-20, 1992, Copenhagen, Denmark, p 327-329

H van Dam, A Mertens and M J S Bellemakers (1988) Effects of liming on attached diatoms in experimental enclosures in an acidified moorland pool (preliminary results) Proceedings Third International Conference on the Conservation an Management of Lakes "Balaton '88' September 11-17, Keszthely, Hungary 51 pp

H van Dam, A Mertens and M J S Bellemakers (1989) Invloed van bekalking van vennen op diatomeeen Diatomededelingen 8 6

H van Dam, A J G Luttikholt, A Mertens and M J S Bellemakers (1990) Effecten van bekalking op diatomeeen in verzuurde vennen op de Tongerense Heide Diatomededelingen 10 15-17

Presentations during (inter)national conferences

M J S Bellemakers Effects of liming on the water quality and vegetation development in shallow acidified lakes and wetlands in the Netherlands Workshop organized by Environmental Canada and the Forest Research Institute of Baden-Wurtemberg Workshop on ecosystem restoration Freiburg, August, 8^{th}-11^{th} 1989

M J S Bellemakers and H van Dam Impact of liming on chemistry, diatoms, macrophytes and moor frogs (*Rana arvalis*) in acid shallow moorland pools in the Netherlands International Conference on Acidic Deposition Its Nature and Impacts, Glasgow, September, 16^{th}-21^{th}, 1990

M J S Bellemakers and M Maessen Restauratie van verzuurde en geeutrofieerde zwak gebufferde ondiepe oppervlaktewateren van laboratorium- tot veldbenadering Studiedag Nederlandse Vereniging voor Aquatische Oecologie Amsterdam (UvA), April, 12^{th}, 1992

M J S Bellemakers, M Maessen and J G M Roelofs Effects of control measurements against acidification on water chemistry and vegetation development of shallow surface waters in the Netherlands The 9th Task Force Meeting of the International Co-operative Programme on Assessment and Monitoring of Acidification of Rivers and Lakes Convention on Long-Range Transboundary Air Pollution, Oisterwijk, November, 2^{th}-4^{th}, 1993

M J S Bellemakers Restoration of the Littorellion vegetation of shallow, slightly buffered, oligotrophic moorland pools Meeting of the Section for Vegetation Research of the Royal Botanical Society of The Netherlands in cooperation with the Department of Plant Ecology and Evolutionary Biology, Utrecht, February, 9^{th}, 1994

CURRICULUM VITAE

Martijn Bellemakers is geboren op 16 juni 1960 te Oss. Na het voltooien van zijn middelbare school (VWO) aan het St. Ursula Lyceum (Horn) ging hij in 1980 biologie studeren aan de Universiteit van Nijmegen. In 1983 werd het kandidaatsexamen met goed gevolg afgelegd, in 1987 werd het doctoraalexamen (met genoegen) afgerond. Direct daarop volgend trad hij in dienst van de afdeling Aquatische Oecologie van Prof. Dr. C. den Hartog als toegevoegd onderzoeker, alwaar de basis werd gelegd voor dit proefschrift. Naast het onderzoek vervulde hij ook onderwijstaken (Oecofysiologie, de veldcursussen Ecologie II en Terschelling en de begeleiding van 35 studenten en iedereen die voor begeleiding/hulp aanklopte). Tevens werden een aantal bestuurlijke activiteiten vervuld (lid vakgroepbestuur Oecologie; computer- en excursiecommissie).
In 1994 en 1995 heeft hij een tweetal projecten uitgevoerd (bij RIKZ Middelburg en Natuurbalans Nijmegen) en per 1 november 1995 is hij als 5de medewerker in dienst getreden van NLnet -later InterNLnet BV- als helpdeskmedewerker. Dit Internet bedrijf is inmiddels uitgegroeid tot een bedrijf van ruim 60 medewerkers, waarbij hij momenteel verantwoordelijk is voor de afdeling bedrijfsvoering en de support afdeling ("goedemorgen, InterNLnet, waarmee kan ik u van dienst zijn?"), een commerciële service voor (eind)gebruikers (telefonisch, e-mail en schriftelijk). Sinds het opzetten van deze support afdeling (1997) maakt hij deel uit van het managementteam van dit bedrijf.
Recentelijk (maart 2000) heeft hij de biologische draad weer opgevat en een groot deel van de eerstejaars colleges Ecologie (community- en systeemecologie) aan de Universiteit van Nijmegen verzorgd.
Naast een lange volleybalcarrière (trainer A NeVoBo) en veel duikervaringen (assistent-instructor PADI, **CMAS) heeft hij in 1994 een oude liefde opgepakt: de hobo. Sinds 1999 speelt hij in het salonorkest van het Nijmeegs Opera- en Operette Gezelschap.

E-mail: Martijn.Bellemakers@interml.net
Dit proefschrift is ook te vinden op: http://web.inter.nl.net/users/Martijn.Bellemakers/these/these.html